# A framework for regulating underwater noise during pile driving

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# Preface

The Vindval research programme is a collaboration between the Swedish Energy Agency and the Swedish Environmental Protection Agency that aims to develop and communicate science-based facts about the impacts of wind power on humans, nature and the environment.

The programme's first two phases in 2005–2014 produced nearly 30 research papers and four so-called synthesis reports. In the synthesis reports, experts compile and assess overall research results and experiences regarding the effects of wind power, both nationally and internationally, in four areas: human interests, birds and bats, marine life and land mammals. The results have provided the basis for environmental impact assessments and for the planning and permit processes associated with wind power installations.

Vindval's third phase, launched in 2014 and ending in 2018, also includes conveying the experience and new knowledge from the wind farms currently in operation. Results from the programme will also be useful in supervisory and monitoring programmes, as well as guidance for government agencies.

As before, Vindval sets high standards for the scientific review of research applications and research results, as well as for decisions on approving the reports and publishing the results.

This report has been written by Mathias H. Andersson, Brodd Leif Andersson, Jörgen Pihl, Leif KG Persson and Peter Sigray from the Swedish Defence Research Agency (FOI), as well as Sandra Andersson, Andreas Wikström, Jimmy Ahlsén and Jonatan Hammar from Marine Monitoring at Kristineberg AB. The authors are responsible for the content, conclusions and recommendations. This report has been translated from the Swedish original (report no 6723, 2016) by Lisa Del Papa, Språkkonsulterna.

Vindval, August 2017

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# Summary – Swedish

Vid byggnation av broar, havsbaserad vindkraft och andra havsbaserade eller strandnära konstruktioner används det oftast någon form av pålningsteknik för att få ner konstruktionen i botten. Detta innebär att ett fundament, balk eller spont hamras eller vibreras ner i botten, vilket kan generera mycket höga ljudnivåer som sprids ut i vattnet och ner i botten. Ljudnivåerna är så höga att marina organismer kan störas, skadas eller till och med dödas.

Idag saknar Sverige fastställda ljudnivåer för när undervattensbuller blir så högt att de kan skada djur i havet. Det saknas därför också vedertagna begränsningsvärden som anger vilka nivåer av undervattensbuller som kan tillåtas för bullrande aktiviteter utan att riskera allvarliga miljöeffekter. Flera länder i Europa har någon form av gränsvärden för när bullernivån under vattnet kan ge upphov till allvarlig miljöpåverkan liksom standarder för hur undervattensbuller skall mätas och rapporteras. Syftet med denna studie var att ta fram ett vetenskapligt underlag rörande ljudet från pålning i havet och dess påverkan på det marina livet. Slutmålet var att utifrån den vetenskapliga information som finns idag ge förslag på ljudnivåer för skador och negativ påverkan som sedan kan användas för att ta fram begränsningsvärden för reglering av undervattensbuller anpassade för svenska vatten och arter. Studien ger ett antal exempel på vilka faktorer som påverkar ljudutbredningen i svenska vatten och hur detta påverkar ett pålningsslags ljudnivå som funktion av avstånd i fyra typområden kring den svenska kusten. Vidare presenteras ingående både tekniska beskrivningar av pålningsaktiviteter, undervattensakustik samt påverkan på marina djur. Denna påverkan (skada och flyktbeteende men ej subtila effekter) demonstreras med hjälp av ett antal typarter som tandvalen tumlare (*Phocoena phocoena*) och fiskarterna torsk (*Gadus morhua*) och sill (*Clupea clupea*) samt fiskägg och fisklarver. I denna studie har författarna gått tillbaka till de originalkällor av information som andra länders gränsvärden grundas på, så att rekommendationerna bygger på vetenskapliga nivåer och inte värden som har avrundats eller på annat sätt ändrats.

Studien presenterar ljudnivåer i tre olika enheter då dessa har olika biologisk relevans för påverkan från en pålningsaktivitet. Inga av dessa värden har frekvensviktats för att anpassas för en specifik art då denna metod ännu inte är helt vedertagen. Den första enheten är ljudtrycksnivå $SPL_{(topp)}$, d.v.s. det maximala över- eller undertryck som den av pålningsslaget genererade ljudpulsen har. Denna enhet har hög relevans för beteendepåverkan. För ljudexponeringsnivå SEL, beräknas ljudnivån över en viss tid och tar då med energin i hela ljudpulsen. SEL är den enhet som visats vara bäst relaterad till hörselskador. $SEL_{(enkel)}$ är värdet för en enkel puls och för det kumulativa $SEL_{(kum)}$ har antalet pulser under en viss tid summerats.

Litteraturstudien på torsk och sill visar att det i dagsläget inte finns några studier som kan användas för att fastställa en artspecifik ljudnivå för skada men litteraturen visar tydligt på att höga bullernivåer kan påverka torsk och

sill negativt. Istället baseras de föreslagna nivåerna i huvudsak på studier på andra arter som har exponerats för pålningsljud i laboratoriemiljö med stöd av studier från mer storskaliga experiment i tankar och hav. De nivåer då fisk riskerar att dödas eller få allvarliga skador på inre organ är $SPL_{(topp)}$ 207 dB re 1 µPa, $SEL_{(enkel)}$ 174 dB re 1 $\mu Pa^2s$ och $SEL_{(kum)}$ 204 dB re 1 $\mu Pa^2s$. Notera att för skada på fisk har det kumulativa värdet högre relevans än enkelvärdet för SEL eftersom studier visar att skador uppkommer efter en viss tids exponering. Nivåerna för påverkan på fiskägg och larver grundas i att inga negativa effekter har observerats vid exponering för ljudtryck från pålning upp till $SPL_{(topp)}$ 217 dB re 1 µPa, $SEL_{(enkel)}$ 187 dB re 1 $\mu Pa^2s$ och $SEL_{(kum)}$ 207 dB re 1 $\mu Pa^2s$. Det finns emellertid mycket få studier relaterat till pålningsljud för dessa livsstadier.

För tumlare finns det fler artspecifika studier gjorda relaterat till buller än för torsk och sill. Det är dock endast ett fåtal som kan användas för att bestämma ljudnivåer som leder till skada eller negativ beteendepåverkan. De ljudnivåer som riskerar ge tillfällig hörselnedsättning (TTS) hos tumlare är $SPL_{(topp)}$ 194 dB re 1 µPa, $SEL_{(enkel)}$ 164 dB re 1 $\mu Pa^2s$ och $SEL_{(kum)}$ 175 dB re 1 $\mu Pa^2s$. Det är framförallt den kumulativa ljudexponeringsnivån $SEL_{(kum)}$ som har stor betydelse för just TTS, dock hänger detta värde ihop med en specifik tid och antalet pulser vilket kan vara svårt att uppskatta i förväg. Vidare avseende permanent hörselskada (PTS) är ljudnivån $SPL_{(topp)}$ 200 dB re 1 µPa, $SEL_{(enkel)}$ 179 dB re 1 $\mu Pa^2s$ och $SEL_{(kum)}$190 dB re 1 $\mu Pa^2s$. Föreslagna nivåer bör uppdateras när nya relevanta forskningsstudier tillkommer.

# Summary

Pile driving is a common technique used during the construction of bridges, offshore wind power, and underwater infrastructure or shoreline structures. It is the process by which a foundation, beam or pole is hammered or vibrated down into the bottom, which can generate extremely loud noise that propagates throughout the surrounding water and sediment. The noise can reach such high levels that marine animals are at risk of disturbance, injury or even death.

Sweden currently lacks established thresholds stating the level at which underwater noise potentially disturbs or injures marine animals. Hence, there are no guidance values for allowable underwater noise levels from noise-producing activities to avoid serious environmental impacts. Several countries in Europe have defined thresholds for when underwater noise can result in severe negative environmental impacts as well as standards for measuring, analysing and reporting underwater noise levels.

The purpose of this study is to review the scientific literature on underwater noise from pile driving and its effects on marine life. The study aims to define the noise levels that can cause injury and other negative effects and, on this basis, recommend noise levels that can be used to establish guidance values for regulating underwater noise for Swedish waters and species. The study presents examples of the factors that contribute to sound propagation in Swedish waters and how this influences the noise level from a pile strike as a function of distance at four study areas along the Swedish coast. Additionally, the study contains a thorough technical description of pile driving activities, basic underwater acoustics and noise effects on marine animals. These effects (injury and behavioral, e.g., flight, but not subtle effects) are demonstrated on representative species such as the harbour porpoise (*Phocoena phocoena*), Atlantic cod (*Gadus morhua*), Atlantic herring (*Clupea harengus*) and on fish larvae and eggs. The study's authors look at the original sources of information that other countries base their guidelines and thresholds on, so the recommendations follow scientifically determined levels rather than values that have been rounded off or otherwise altered.

The study presents sound levels in three different units, each with different biological relevance to the effects caused by a pile driving activity. None of the sound levels have been frequency weighted for a specific species, as this method is not yet fully established. The first unit used is the sound pressure level $SPL_{(peak)}$, which is the maximum overpressure or underpressure of the noise pulse generated by the pile strike. This unit has a high relevance for behavioural effects. The sound exposure level, SEL, is the calculated energy level over a period of time and expresses the energy of the entire sound pulse. SEL is the unit most related to hearing impairing effects. $SEL_{(ss)}$ is the value for a single strike while $SEL_{(cum)}$ is the cumulative value of a determined number of pulses over a period of time.

The review revealed that for Atlantic cod and Atlantic herring there are currently no studies that can be used to determine a species' specific threshold value for injury, but studies show that loud noise can affect both species negatively. Because of this, the recommended noise levels for injury are based mainly on studies on other species exposed to pile driving noise in laboratory environments, supported by studies conducting large-scale experiments in tanks and oceans. The levels at which fish are at risk of death or sustaining serious injury to internal organs is $SPL_{(peak)}$ 207 dB re 1 µPa, $SEL_{(ss)}$ 174 dB re 1 µPa$^2$s and $SEL_{(cum)}$ 204 dB re 1 µPa$^2$s. Note that for injury in fish, the cumulative sound exposure level has higher relevance than the single-strike level as the cited studies found injuries after a certain time period of exposure. The thresholds for fish larvae and eggs are based on the fact that no negative effects were observed at exposures of up to $SPL_{(peak)}$ 217 dB re 1 µPa, $SEL_{(ss)}$ 187 dB re 1 µPa$^2$s and $SEL_{(cum)}$ 207 dB re 1 µPa$^2$s. However, there are relatively few studies on early life stages of fish.

There are more species-specific studies on harbour porpoises regarding noise than there are for Atlantic cod and Atlantic herring. Nonetheless, only a few can be used to determine thresholds that will lead to injury or negative behavioural effects. The levels at which there is a risk of a temporary impact on hearing, i.e. temporary threshold shift (TTS), for the harbour porpoises is $SPL_{(peak)}$ 194 dB re 1 µPa, $SEL_{(ss)}$ 164 dB re 1 µPa$^2$s and $SEL_{(cum)}$ 175 dB re 1 µPa$^2$s. When it comes to TTS, the cumulative sound exposure level, $SEL_{(cum)}$, is of primary importance. However, this unit is dependent on a specific time and number of pulses. For permanent threshold shift (PTS), the level is set to $SPL_{(peak)}$ 200 dB re 1 µPa, $SEL_{(ss)}$ 179 dB re 1 µPa$^2$s and $SEL_{(cum)}$ 190 dB re 1 µPa$^2$s. The recommended level should be revised as new relevant studies are conducted.

# Glossary

| | |
|---|---|
| ACCOBAMS | Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area. |
| ASCOBANS | Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas. |
| Acoustic impedance | The ratio (the resistance) of sound pressure to the particle velocity of the sound wave. |
| CTD | (Conductivity, Temperature and Depth). Sensor that measures water conductivity (for the calculation of salinity), temperature and depth. From these values, the sound velocity profile can then be calculated. |
| Cut-off frequency | In underwater acoustics, the cut-off frequency that sets the limit for the lowest frequency that can propagate in shallow waters. |
| Far field | The noise at a distance of at least 10 wavelengths away from the source (for the lowest frequency). An example is the wavelength at the frequency 100 Hz about 15 metres in the water. At this distance, one can make a linear adjustment of sound pressure as a function of distance. |
| Full-field model | Numerical model that calculates the exact solution to the elastodynamic wave equation everywhere, that is, in both the near and far fields. |
| HELCOM | The Helsinki Commission. Convention on the protection of the Baltic Sea's marine environment. |
| ICES | International Council for the Exploration of the Sea. |
| IUCN | International Union for Conservation of Nature. |
| Near field | Can be defined in different ways. In this report, it refers to the sound field near the source, for example within 10 wavelengths from a source. |
| Numeric model | An algorithm implemented in a computer in order to solve a mathematical problem. |
| OALib | Ocean Acoustics Library (oalib.hlsresearch.com). Website that provides software and data for modelling sound propagation in water. Funded by the U.S. Office of Naval Research. |
| OSPAR | The Convention for the Protection of the Marine Environment of the North-East Atlantic. |

| | |
|---|---|
| PSU | Practical salinity unit. A dimensionless unit used to estimate the salinity of water. Salinity is the concentration of salt in water. PSU is obtained by converting the conductivity of water to salinity and is expressed in parts per thousand. |
| Peak | (P) Shortened to (peak) in this document. Also called zero-peak (Lz-p). Is expressed in dB re 1 µPa. |
| Peak-to-Peak | (P-P) Shortened to (peak-peak) in this document. Is expressed in dB re 1 µPa. |
| PTS | Permanent threshold shift. A permanent hearing injury that involves a reduced ability to hear sounds within the damaged frequency range. |
| RMS | Root-mean-square is the same as the effective value. The square root of the average of the square of the sound pressure over a given duration. |
| SEL | Sound exposure level. Is expressed in dB re 1 $\mu Pa^2 s$. Can be given for both a single pulse, $SEL_{(ss)}$, and as a weighted mean over many pulses, $SEL_{(cum)}$. |
| Sound velocity profile (SVP) | The velocity of sound as a function of water depth. |
| Source level (SL) | Same as signal strength. Refers to a reference intensity or reference effect generated by a plane wave with the sound pressure re 1 µPa RMS at a distance of 1 metre in an isotropic water volume with density 1,000 $kg/m^3$ and sound speed 1,500 m/s. |
| SPL | Sound pressure level. Is expressed in dB re 1 µPa in water and dB re 20 µPa in air. |
| Stratification | In geology, the order in which the soil or rock layers follow each other. |
| Third-octave band | Division of an octave band in three parts. Also known as a 1/3 octave band. An octave band is when the ratio between the lowest and highest tone is 2:1. |
| TL | Transmission loss. Indicates how much a sound wave weakens from a point situated 1 metre from the sound source to a point at a distance R. TL consists of three parts, geometric dispersion, absorption and anomaly. |
| TTS | Temporary threshold shift. A temporary hearing injury that involves a reduced ability to hear sounds within the damaged frequency range. |

# 1 Background

During the construction of bridges, offshore wind farms and other offshore or near-shore structures, some form of pile driving method is often used to drive the structure into the bottom. This means that a foundation pile, beam or pole is hammered or vibrated down into the bottom. These operations can generate extremely high noise levels that propagate into the water and downward to the bottom. Sound can travel very far and fast in water, about four times faster than in air due to the higher density of water. This noise is so high that marine animals can be disturbed, harmed or even killed. In addition, the frequency content of the emitted sound coincides with many marine organisms' zone of audibility (Figure 1). Much construction is currently taking place in Europe's maritime and coastal regions, with more planned in the future. In Sweden, few offshore wind farms have yet to be created despite the availability of the required permits. However, work on bridges and in ports takes place relatively often. The body of knowledge on how this noise impacts marine life has been expanded in recent years, with much measurement data primarily from the establishment of offshore wind farms. But critical gaps in knowledge remain.

Today, Sweden lacks established maximum noise levels for indicating when underwater noise can begin to cause serious environmental effects, similar to those found on land. Several European countries have some form of thresholds or guidance values indicating when serious environmental effects can occur, as well as standards for measuring and reporting underwater noise.

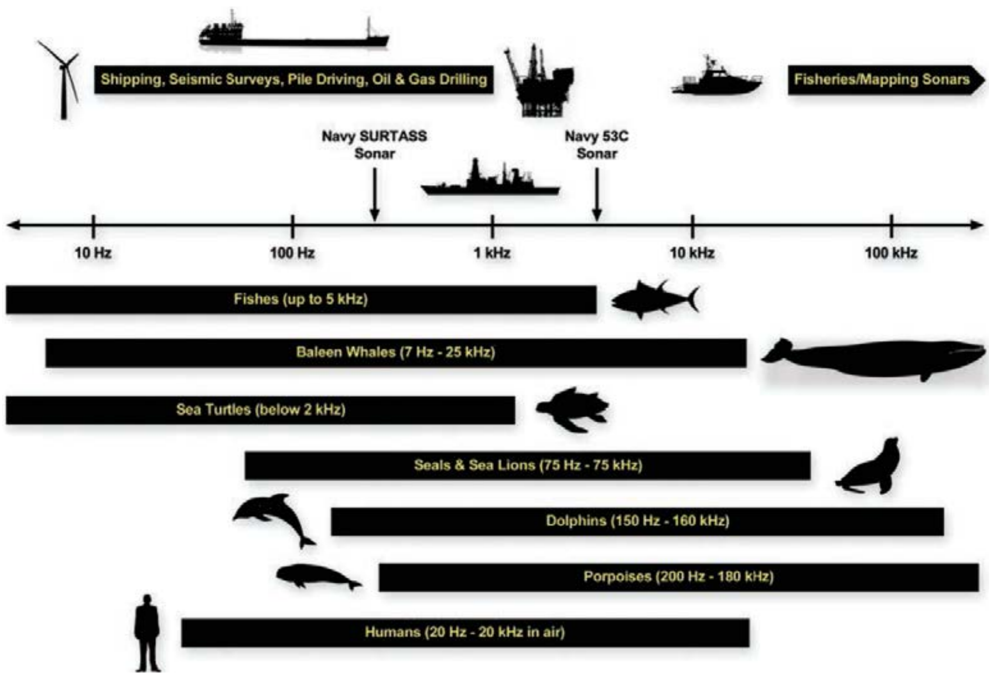



Figure 1. Overview of the overlap in frequency between human activities and the audibility zones of marine animals, modified after Scholik-Schomer (2015).

Underwater sound is one aspect of good environmental status according to the EU's Marine Strategy Framework Directive and thereby part of Sweden's Marine Environmental Ordinance, the national implementation of this directive. This ordinance (HVMFS 2012:18) specifies two criteria for good environmental status, one of which one concerns impulsive sound sources. There are still no established threshold values for impulsive sound, nor are there any environmental quality standards with indicators for achieving good environmental status by 2020.

This is because the body of knowledge on sound levels in Swedish marine areas is relatively low, and the impact on population levels and ecosystems is not yet fully understood (Swedish Agency for Marine and Water Management, 2015). Environmental quality standards for marine environments are based on Chapter 5 of the Swedish Environmental Code and apply for inspections and certifications, which are important tools for protecting the marine environment. The effects of underwater noise and the standards for protection measures are also part of the Swedish Environmental Code (Chapter 2), requirements for environmental assessment (Chapter 6), provisions for water operations (Chapter 11) and regulatory requirements (Chapter 26). The monitoring programme for the marine environment directive contains a plan for future monitoring of underwater noise. This plan also states that a register should be kept containing the time and location of activities like pile driving that have generated high impulsive sounds in Swedish waters (Swedish Agency for Marine and Water Management, 2015).

## 1.1 Underwater sound

In water, sound energy propagates as particle motion. This motion creates longitudinal pressure variations in which the medium is compressed and decompressed, giving rise to sound pressure fluctuations. Pressure and motion are related to each other through acoustic impedance. There are several fundamental differences between these two phenomenons, one being that particle motion contains information about sound direction. Furthermore, transmission losses (the dampening of sound) are different for pressure and acceleration in the vicinity of a sound source, near the seabed and near the surface, which complicates the calculations. Sound pressure fluctuations are expressed in the unit of pressure Pascal (Pa); for historical and practical reasons, the sound pressure level in air is referenced to 20 µPa and in water to 1 µPa. The logarithmic scale in decibels (dB) has been introduced in acoustics because of its great dynamics. This makes it important to specify which reference is used when specifying a sound pressure level in decibels: 1 or 20 µPa. So, one cannot directly compare decibel values measured in air with values measured for water (see Vindval reports Sigray et al. (2009), Andersson och Sigray (2011) and Andersson et al. (2011) for a further description of underwater sound and effects from other types of sound sources).

Understanding that sound manifests itself as both pressure and particle motion significantly aids the understanding of environmental impact on marine animals, who have different sensor systems for detecting sound. All fish can detect particle motion with their inner ears, which contain ear stones (otoliths) and a lateral line organ. Even invertebrates like squid can detect particle motion, but this animal group does not fall within the scope of this study. Fish that have an air-filled cavity in their body, such as a swim bladder, can transform sound pressure to motion and thus increase their sensitivity in terms of both frequency and level. Marine mammals can only register sound pressure with their ears. Chapter 6 discusses this topic further and describes the hearing of harbour porpoises, herring and cod in more detail.

Most studies about the effects of sound on marine animals only discuss sound pressure. But it is generally accepted that particle motion also plays a significant role for the effects of pile driving noise, especially for fish without swim bladders and for demersal (bottom-feeding) fish (Mueller-Blenkle et al., 2010; Van der Graaf et al., 2012; Popper et al., 2014). This is because impact pile driving can generate very high levels of particle motion in the water (Thomsen et al., 2015) and at the surface of the seabed (Miller et al., 2015; Hazelwood and Macey, 2015). There are currently no national or international guidelines on how to measure or calculate particle motion. Therefore, particle motion is not further addressed in this report, although it is an area that should be studied in order to understand the full impact of pile driving noise on fish and invertebrates.

## 1.2 Purpose of the study and reader's guide

The purpose of this study is to present scientific documentation on underwater pile driving noise and its effects on marine life. The ultimate goal is to establish guidance levels for noise that causes injury and other negative effects, based on the current scientific information, which can then be used to determine thresholds for regulating underwater noise adapted to Swedish waters and species. Chapter 2 contains the noise levels that this study found to be harmful to Swedish species and marine areas. It also contains examples that illustrate sound propagation in Swedish waters, and how the local acoustic environment affects sound levels from a pile driving activity as a function of distance in four study areas around the Swedish coastline. For the sake of brevity, Chapter 2 contains summary information that is not further described in detail. Subsequent chapters contain in-depth information such as technical descriptions that offer the reader insight into pile driving activities, underwater acoustics and effects on marine life. The study has returned to original sources used by other countries to determine their own thresholds. This was done in order to provide recommendations grounded in scientifically established levels. The levels have not been rounded off or otherwise changed.

Chapter 3 discusses the nature of pile driving noise and previously measured noise levels as well as measuring standards for underwater sound.

Chapter 4 describes the factors affecting sound propagation in Swedish waters, and Chapter 5 describes the noise mitigation methods in use today and how they can reduce the impacted area. Chapter 6 describes the influence on marine organisms that can result in direct and lasting effects, such as injury and flight behaviour on an individual level. The species and their scientific names that are discussed include the harbour porpoise (*Phocoena phocoena*) and the fish species herring (*Clupea clupea*) and cod (*Gadus morhua*), as well as fish eggs and larvae. Thresholds and guidelines from other countries relating to underwater noise are described in Chapter 7.

# 2 Recommendations on harmful noise levels for pile driving

## 2.1 Definition of threshold

A threshold indicates the level of a harmful substance or activity that must not be exceeded by a noise-producing activity. A threshold should be possible to manage from an administrative point of view, be measurable and at the same time have high biological relevance. In other words, it should be able to be linked to the percentage of a population that could be exposed to harmful noise and to how vulnerable the population is. The spatial estimate depends, however, on a number of local environmental parameters that affect sound propagation; see Section 2.4. Several countries have up to three different types of thresholds for underwater noise from impact pile driving, depending on which environmental impacts require prevention. For a detailed description of the physical units that relate to the harmful levels recommended below and how these are related to injury, see the in-depth literature survey in subsequent chapters. The levels recommended in this study can be used by government authorities to determine which noise levels are acceptable and which should not be exceeded. When multiple noise levels are indicated, the level that is exceeded first is the one that applies. The distance from a pile driving activity at which these noise levels can occur cannot be generalised because the level and the local environmental conditions of the sound source play a large role for the radiating sound and how far it will propagate. To estimate these parameters, modelling and direct measurements can be carried out at the permit application stage so as to detect any environmental risks early on in a permit process.

## 2.2 Recommended noise levels

Recommended noise levels for underwater pile driving that risk resulting in serious environmental impacts on fish, fish eggs and larvae are presented in Table 1 and for harbour porpoises in Table 2. The reasoning behind using these levels is briefly explained in this section, and an in-depth discussion is given in the next chapter. Note that the noise levels for TTS are not weighted, i.e., they are not adapted to the hearing ability of the porpoises (see explanation in Section 3.3). At present, weighting curves for fish are lacking. In some cases, they are available for harbour porpoises but the methodology is not standardised and there is uncertainty around how to make this adjustment for different types of sound sources (Tougaard et al., 2015). No European country currently uses weighted threshold values for pile driving noise, although in 2015 the U.S. National Oceanic and Atmospheric Administration (NOAA) proposed both weighted and unweighted thresholds for underwater noise. The proposed noise levels presented here are not weighted because there is no standardised method for doing this. However, a weighting methodology should be included in future threshold studies when relevant research results become available in this field.

There are three different units for each type of impact and animal group. Multiple units are used because noise can affect animals differently (Southall et al., 2007), and because a measured sound level can be presented in several ways. Sound pressure level ($SPL_{(peak)}$) is the maximum overpressure or underpressure exhibited by the sound pulse generated by one pile strike. This value is easy to measure, and biological relevance is high since several scientific studies have noted a correlation between SPL and behavioural impacts – and even physiological damage.

Sound exposure level (SEL) is calculated over a given period of time and takes into account the energy of the entire sound pulse. SEL is the unit best used to express hearing injury thresholds in harbour porpoises (Tougaard et al., 2015) and injury to fish (Halvorsen et al., 2011, 2012a, b; Casper et al., 2012, 2013). $SEL_{(ss)}$ is the value of a single strike, and $SEL_{(cum)}$ expresses the cumulative value of several strikes over a given period of time. These two SEL units have different uses and are related to injury in various ways. $SEL_{(ss)}$ is easy to measure and has good relevance to injury in animals. However, it is not a metric that can describe the total received energy level for an animal. $SEL_{(cum)}$ gives us an idea of how much energy a given stationary point at a certain distance has received. But making the link to an animal's received sound level is difficult, since many animals can travel and be exposed to more or fewer sounds (this reasoning does not apply to eggs and larvae and many sessile invertebrates). Studies show that it can be appropriate to have several different thresholds since both the number of pile strikes and the levels will influence the extent of the injury. Different thresholds can thus be used at the same time when prescribing the limiting conditions for an offshore wind farm operation. As regards the SEL unit, it should be noted that few strikes with high $SEL_{(ss)}$ give the same $SEL_{(cum)}$ as many strikes with low $SEL_{(ss)}$ (Halvorsen et al., 2011, 2012a).

### 2.2.1 Fish

Based on the literature, it is observed that high noise levels can adversely affect both cod and herring. However, there are too few studies on cod and herring that can be used to determine species-specific noise levels that can be harmful. The proposed levels are instead based on studies of other species that have been exposed to pile driving noise in a laboratory environment (Halvorsen et al., 2011, 2012a, b; Casper et al., 2013) (Table 1). The authors of these studies suggest guidelines for noise levels where injury occurs corresponding to $SEL_{(cum)}$ 207 dB re 1 μPa²s, which was reached when the fish were exposed to 960 sound pulses with a sound exposure level of $SEL_{(ss)}$ 177 dB re 1 μPa²s. The injury that occurred is considered to be of such magnitude that it can affect the fishes' survival. However, injury to internal organs was observed at lower noise levels equivalent to $SEL_{(cum)}$ 204 dB re 1 μPa²s, since the fish were exposed to the same number of pulses but with a sound exposure level of $SEL_{(ss)}$ 174 dB re 1 μPa²s. The fish are expected to recover from these injuries in favourable environments without predators, and with the right current conditions and easy access to food (Popper et al., 2014). Most fish do not live under these conditions, and in the current situation this lower noise level is instead proposed as a level for injury in fish (see a more detailed description of the results from the various studies on which the proposed noise levels are based in Section 6.2.4). Note that for injury to fish, the cumulative value has higher relevance than the single value for SEL because the studies present the injury after a certain period of exposure. It remains unclear how many pulses and how long fish can be exposed before injury occurs. The proposed $SPL_{(peak)}$ value (207 dB re 1 µPa) is taken from the guidelines in Popper et al. (2014), based on the same laboratory studies.

**Table 1. Recommendations on thresholds for pile driving noise for fish, fish eggs and larvae. The threshold levels are presented as SPL (maximum overpressure or underpressure of the generated sound pulse), $SEL_{(ss)}$ (sound exposure level for a single sound pulse) and $SEL_{(cum)}$ (sum of the sound exposure levels for a number of pulses in a given period of time). See the text for the reasoning behind using these levels.**

| | Fish | Eggs and larvae |
|---|---|---|
| Mortality and injury to internal organs | $SPL_{(peak)}$ 207 db re 1 µPa | $SPL_{(peak)}$ 217 dB re 1 µPa |
| | $SEL_{(ss)}$ 174 dB re 1 µPa²s | $SEL_{(ss)}$ 187 dB re 1 µPa²s |
| | $SEL_{(cum)}$ 204 dB re 1 µPa²s | $SEL_{(cum)}$ 207 dB re 1 µPa²s |

Although the recommended noise level for injury in fish is based on studies of multiple species with anatomical and morphological differences, a degree of uncertainty still remains. There are relatively few species that have been tested, and the equipment allows for only smaller fish to be exposed. Conditions in the laboratory also differ from the fish's natural habitat. It is primarily the exposure time that is expected to be lower for the wild fishes that are able to escape the harmful noise levels. Note that the injury observed at $SEL_{(cum)}$ 204 dB re 1 µPa²s occurred when fish were exposed to 960 sound pulses ($SEL_{(ss)}$ 174 dB re 1 µPa²s), which is the equivalent exposure for about 24 minutes. Depending on the individual's size and the different conditions in its surroundings, a cod can swim approximately 550–1,300 m in this time (Beamish, 1966; Wardle, 1977; Thurston and Gehrke, 1993) and a herring can swim the equivalent of approximately 1,500 m (He and Wadle, 1988; He, 1993). A flight behaviour assumes, however, that the fish react to the noise. During exposure to pile driving noise, behavioural changes were observed in cod in large-scale experiments in the sea at $SPL_{(peak)}$ 140 to 160 re 1 µPa dB (Mueller-Blenke et al., 2010) and in European sprat (similar hearing to herring) at $SPL_{(peak\text{-}peak)}$ 163 re 1 µPa and $SEL_{(ss)}$ 135 dB re 1 µPa²s (Hawkins et al., 2014). The results show that the fish react to the pile driving noise and are thus expected to swim away from the noise. At the same time, there are studies indicating that fish exposed to high noise levels remain within an area if it is important enough to the fish's survival or reproduction (Wadle et al., 2001; Pena et al., 2013).

At present, no noise levels are proposed for flight behaviour or a temporary threshold shift (TTS) in fish. This is because unlike physiological damage to internal organs, both flight behaviour and hearing damage are linked to the species' specific sensitivity to frequency and sound intensity. And using the existing literature, it is not possible to assess whether flight behaviour negatively affects the species at the population level or whether the effect of the impact is related to the area and period of time.

### 2.2.2 Fish eggs and larvae

The proposed noise levels for damage on fish eggs and larvae (Table 1) are based on the fact that no adverse effects were observed at exposures to pile driving-induced noise levels up to $SEL_{(cum)}$ 207 dB re 1 μPa²s (100 strikes), $SEL_{(ss)}$ 187 dB re 1 μPa²s and $SPL_{(peak)}$ 217 dB re 1 μPa (Bolle et al., *submitted manuscript, b*). However, several studies have observed increased mortality due to the noise from airguns at $SPL_{(peak)}$ 217 dB re 1 μPa and above (see Table 10). The immobility of fish eggs and larvae means that they experience a longer exposure than a larger fish. Harmful noise levels occur only near the sound source, and because the eggs and larvae exhibit naturally high levels of mortality in the wild, several authors point out that mortality caused by high impulsive noises is assessed to be insignificant for the population.

### 2.2.3 Harbour porpoises

For harbour porpoises, there are more species-specific studies related to noise than for cod and herring. However, only a few can be used to propose harmful noise levels. The proposed thresholds for harbour porpoises presented in Table 2 are based on existing literature on harbour porpoises and on international thresholds. For single pulses ($SPL_{(peak)}$ 194 dB re 1 μPa and $SEL_{(ss)}$ 164 dB re 1 μPa²s ), the noise level that causes TTS is based on a study by Lucke et al. (2009), which is deemed to have the highest relevance in relation to a pile driving noise out of all published assessments. The published study presents a peak-peak value which here has been recalculated to a peak value (–6 dB; see Section 3.2.3). The noise level for the cumulative sound exposure level ($SEL_{(cum)}$ 175 dB re 1 μPa²s) corresponds to the Danish threshold value (Tougaard, 2015), which is based on a study by Kastelein et al. (2015) (see Section 7.3.1). The cumulative sound exposure level has major significance for TTS. This value is, however, linked to a specific duration and number of pulses. In this case, $SEL_{(cum)}$175 dB re 1 μPa²s is related to one hour's pile driving with 2,760 pulses. Such exposure assumes that the harbour porpoise does not move away from the source of disturbance, which cannot be considered expected behaviour in a natural environment. However, there are other combinations of sound level and number of strikes over time that can result in this harmful level.

**Table 2. Recommended noise levels for pile driving noise that can result in temporary (TTS) or permanent (PTS) hearing injury in harbour porpoises. The levels are presented as SPL (maximum overpressure or underpressure of the generated sound pulse), SEL (sound exposure level over the entire pulse) and $SEL_{(cum)}$ (sum of the sound exposure levels for a number of pulses in a given period of time). See the text for the reasoning behind using these levels.**

| TTS | PTS | |
|---|---|---|
| Harbour porpoise | $SPL_{(peak)}$ 194 dB re 1 μPa | $SPL_{(peak)}$ 200 dB re 1 μPa |
| | $SEL_{(ss)}$ 164 dB re 1 μPa²s | $SEL_{(ss)}$ 179 dB re 1 μPa²s |
| | $SEL_{(cum)}$ 175 dB re 1 μPa²s (≥1 h) | $SEL_{(cum)}$ 190 dB re 1 μPa²s (≥1 h) |

No measured thresholds exist for permanent threshold shift in the harbour porpoise because it is unethical to expose them to such high noise levels. There is, however, a study on a closely related species of porpoise, the Asian porpoise (*Neophocoena phocaeenoides*) (Popov et al., 2011) that is relevant for PTS in harbour porpoises. Since newer studies show a frequency dependence in marine mammals and since the study is not considered representative of pile driving noise, PTS is instead calculated for single pulses ($SEL_{(ss)}$ 179 dB re 1 µ$Pa^2$s) on the basis of the study by Lucke et al. (2009) (TTS + 15 dB). This reasoning is presented in the revision of the Danish threshold values (Tougaard, 2015). The proposed noise level causing PTS from cumulative sound exposure is, as for TTS, based on a study by Kastelein et al. (2015) and calculated to $SEL_{(cum)}$ 190 dB re 1 µ$Pa^2$s (TTS + 15 dB) in the Danish threshold values. The proposed $SPL_{(peak)}$ value (200 dB re 1 µPa) for PTS is taken from a line of reasoning in NOAA (2015) and the study by Lucke et al. (2009) (TTS + 6 dB), but should be used with caution since it is the SEL value that has the best scientific agreement in the literature with regard to the impact on the hearing of marine mammals (Southall et al., 2007; Finneran, 2015).

The scope of this study did not include to propose noise levels for avoidance or flight behaviour in harbour porpoises. But the levels currently used and presented in Danish and Dutch guidelines for a behavioural impact are $SEL_{(ss)}$ 140 dB re 1 µ$Pa^2$s, which are based on a study by Dähne et al. (2013).

## 2.3 Methods for reducing the environmental impact of pile driving noise

To get an idea of the noise levels a planned activity might generate, sound propagation calculations can be made for the specific area. These calculations can also be verified by measuring the actual sound propagation in the area. This work provides important information which in turn can be used in a risk assessment for the underwater noise generated by the planned activity. Furthermore, one should estimate the biological values in the area and the periods during the year that are most critical. If the dates of the activities and the biological values overlap and the potential risk of injury become too high, several options are available for reducing these risks. The generated noise can be reduced by using various pile driving techniques, and the radiated noise can be mitigated with mitigation systems such as bubble curtains or isolation casings (see Chapter 5). Animals – primarily seals and harbour porpoises – can also be scared away from the area closest to the installation where they risk being harmed by using acoustic deterrent devices (ADDs) (Kyhn et al., 2015; Mikkelsen et al., 2015). Some fish species have even been shown to respond to sounds similar to those generated by ADDs (Kastelein et al., 2007, 2008). The operator can also create a similar effect by using a ramp-up method, in which they gradually increase the driving rate at the start of an activity. If the latest mitigation techniques and preventive measures such as ADDs are used, the impact zone for injury to animals around a pile can be limited to a few hundred metres or kilometres from the pile driving activity; see Section 5.5 and Bellman et al. (2015).

These measures are commonly used in the pile driving guidelines of several European countries. Installation work can also be appropriately scheduled in order to avoid sensitive periods of spawning and mating.

The better data there is for biological values and the noise levels that are generated and radiated, the easier it will be to prevent the negative environmental impact of noise. There may be times when no noise mitigation measures are necessary, perhaps because an area lacks biological values or because the sound propagation conditions are such that noise levels will not cause harm within a larger area. Yet there may be occasions when noise mitigation measures must be used, as when the potential injury risks becoming extensive or when installation work is done during a specific critical period from a biological point of view.

## 2.4 Towards regulation of underwater noise

This report can be used by operators and industry as a scientific basis for reducing the effects of impact pile driving on marine animals. After this report, based on the guidance levels, the next step is determining the prescribed thresholds for pile driving activities and the approach that will ensure these values are not exceeded. The following list summarises the conclusions in this report and contains recommendations on which information to include in future regulatory documents:

- Recommended harmful levels for injury to harbour porpoises, fish, fish larvae and eggs can be converted to threshold values straight off or with adjustments made based on the precautionary principle as relates to the current body of knowledge.
- When developing national thresholds, the most appropriate units for the current environmental impact should be used, i.e. $SPL_{(peak)}$, $SEL_{(ss)}$, or $SEL_{(cum)}$. When multiple noise levels for a threshold are indicated, the level that is exceeded first is the one that should apply.
- In the early stages of the application process, the operator should estimate the probable source level for the current activity and how this sound might propagate in the relevant area during the months when installation is scheduled. This is best done through a combination of actual measurements and modelling. A threshold value can thus be connected to an area around the activity in order to estimate the spatial impact.
- An assessment should be made of the biological values and the biologically critical periods in the area affected by an installation in order to link the spatial impacted area to the biology.
- The population densities of the species (number of individuals per surface area) and their vulnerability should be taken into account when determining the threshold values within the exposed area.
- The operator should propose appropriate measures for reducing the potential negative effects, such as mitigation techniques, acoustic deterrent devices, and choice of installation period.

- The operator should perform a survey when construction work is started to ensure that the established thresholds are not exceeded.
- Standards should be established for taking surveys, managing and analysing data, and documenting requirements. Since the first version of this report, an ISO standard has been published called ISO 18406:2017 "Underwater acoustics – Measurement of radiated underwater sound from percussive pile driving".

## 2.5 Sound propagation from pile driving in Swedish waters

To give an idea of the distances at which the recommended noise levels can arise from a pile driving activity, $SEL_{(ss)}$ has been calculated in four selected areas. These areas were chosen because wind farms exist in the area and because they contain designated areas for offshore wind power or areas where one or more installations are under consideration. In addition, the sites represent areas with varying acoustic environmental parameters such as bottom type and salinity. There are, however, great uncertainties associated with the parameters within each area. The examples below should not be viewed as "exact" but instead illustrate the variations exhibited by the sound propagation.

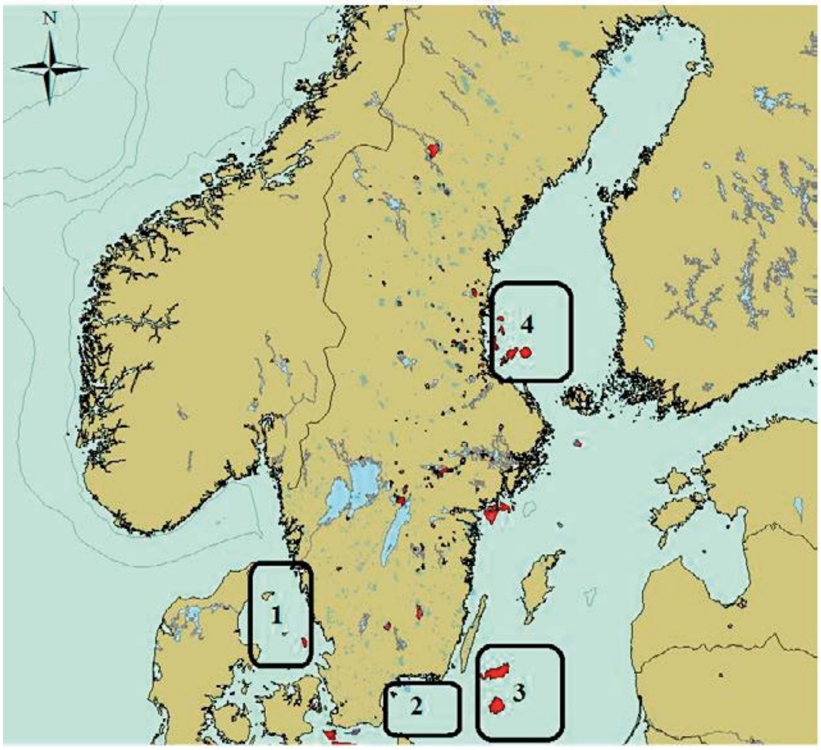



Figure 2. Map of the selected sea areas where sound propagation from a pile strike is calculated. 1 Kattegat, 2 Hanöbukten, 3 The Southern Baltic and 4 The Bothnian Sea. Coloured polygons in the sea are areas designated as interesting for renewable energy production at the national level.

Examples of relatively extreme, but not rare, sound propagation conditions has been used. The four selected areas are the Kattegat, Hanöbukten, the Southern Baltic Sea and the Gulf of Bothnia (Figure 2). The sound source used in the calculations has the same sound level as that described in Section 4.3, i.e., an unmitigated pile driving operation in the German North Sea. The depth at the German measurement site was 20 m and the steel piles were 6 m in diameter. The impact energy level was 700 kJ and the pile was driven into a sandy seabed. The measurement was made 750 m from the pile driving. The estimated equivalent source level was $SEL_{(ss)}$ 226 re 1 μPa$^2$s for the frequency range of 10 Hz to 25 kHz (see 4.3.1). Based on these parameters, acoustic modelling was done using both a ray-tracing based model for high frequencies and a full-field wave-equation based model for lower frequencies (below 800 Hz). When mitigation techniques are used during a pile driving, they will have a positive effect on the noise level and will reduce the radiated noise. This means that the equivalent source level is adjusted downwards, thereby reducing the radiated intensity. To obtain an approximate figure for the levels, the number of decibels used to mitigate the radiated noise is deducted from the table values below. Today, the sound level has been successfully reduced by 10–20 dB (see also Chapter 5).

Sound velocity profiles are based on climate data from the Swedish Meteorological and Hydrological Institute, and should be viewed as type values for the selected months of February and August and for the area as a whole. In fact, a sound velocity profile can vary significantly over time. The high-resolution bottom data were taken from the Geological Survey of Sweden and are meant to resemble a type bottom (i.e. berock, sand, gravel, clay) for that specific marine area. The reference bottom type selected for use in the calculations is the one among the different bottom types in the area that is considered to be the most common. The water depth for each area was taken from a Swedish Maritime Administration database.

Estimated noise levels are highly dependent on the environmental parameters used in the model. Sound characteristics for the bottom sediment are often unavailable. The same type of bottom can have different acoustic characteristics. In the absence of locally specific data, the modelled sound propagation should be regarded as guidance. The measured values might differ from the modelled ones. To make the results more accurate, the local acoustic characteristics should be established. In these examples, the unit $SEL_{(ss)}$ is used to illustrate the received noise levels as a function of distance. This is because the data were delivered in this unit from Germany and because this unit is suitable for studying the effects on the hearing of harbour porpoises. The unit can also be used to study injury to fish. But it should be noted that the onset of injury also depends on the number of sound pulses over time. For example, in the case of fish the recommended noise levels $SEL_{(ss)}$ 174 dB re 1 μPa$^2$s are based on an exposure of 24 minutes (960 pulses) during which the fish were not able to move.

### 2.5.1 Kattegat

In the Kattegat, it is assumed that the bottom consists of a 0.5 m thick sediment layer of watery clay on top of a 20 m thick sediment layer of sand underlain by crystalline bedrock. The salinity was set at 34 PSU. The calculations were made out to a distance of 25 km. The water depth in the area along a bearing of 45° to 225° ranged between 20 and 40 m. Sound velocity curves that represent a typical February (winter) and August (summer) were used (Figure 3, left). August exhibits a downward-refracting profile, which means that the sound is refracted down to the seabed, which in turn absorbs a large part of the sound. During February, sound was refracted from the surface down to a 20 m depth up to the surface. The surface reflects the sound, which then becomes locked in a channel. During a pile driving operation with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 μPa²s, the sound will travel longer in February than in August (Figure 3, right). The phenomenon becomes apparent at a distance of about 3 km, at which the two curves diverge. This sound propagation can be considered rather extreme – but not rare – in the area. The two vertical lines indicate two of the recommended harmful noise exposure levels for fish and harbour porpoises, respectively. It should be emphasized that if the source level changes, the blue line and the red line shift vertically. Table 3 shows the received sound exposure levels at different distances when the equivalent source level was set to $SEL_{(ss)}$ 226 dB re 1 μPa²s.

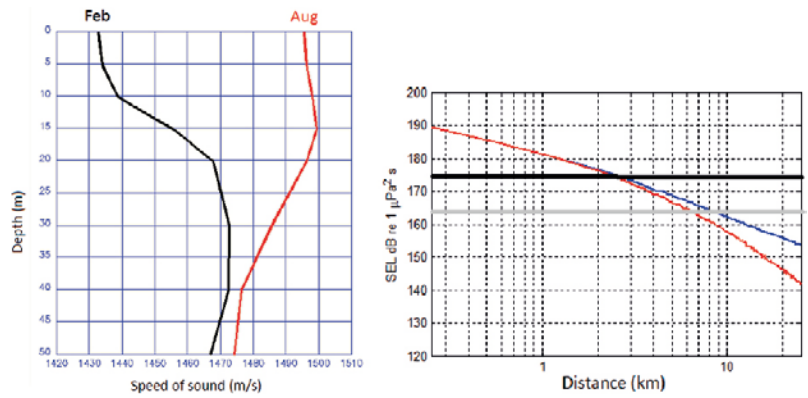



Figure 3. Left: Sound velocity profiles for February (black) and August (red) in the Kattegat. Right: The sound propagation for a pile strike with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 μPa²s as a function of distance along the bearing of 45° to 225° in the Kattegat. The calculations were made with sound velocity profiles typical for February (blue) and August (red). The grey line shows the recommended threshold for TTS for harbour porpoises ($SEL_{(ss)}$ 164 dB re 1 μPa²s), and the black line mortality and injury to the internal organs of fish ($SEL_{(ss)}$ 174 dB re 1 μPa²s). Note that the injury also depends on the number of sound pulses over time. The figure shows examples of modeled sound propagation with special sound propagation conditions and assumptions about exposure (see explanation in Section 2.5).

**Table 3. Sound exposure level as a function of distance in the Kattegat when the equivalent source level was set at $SEL_{(ss)}$ 226 dB re 1 μPa²s.**

| Distance from source (km) | $SEL_{(ss)}$ (dB re 1 μPa²s) FEB. | $SEL_{(ss)}$ (dB re 1 μPa²s) AUG. |
|---|---|---|
| 0.75 | 183 | 183 |
| 1.5 | 179 | 178 |
| 3 | 169 | 172 |
| 5 | 157 | 166 |
| 10 | 162 | 158 |
| 20 | 155 | 146 |

### 2.5.2 Hanöbukten

In Hanöbukten, the bottom is assumed to consist of a 0.5 m thick layer of clay on top of 3 m of moraine underlain by limestone. The salinity was set at 8.6 PSU. The calculations were made out to a distance of 25 km. The water depth in the area along a bearing of 90° to 270° ranged between 50 and 60 m. Source depth was set at 32 m. The sound velocity profiles show great differences from the surface down to 30 m because the surface waer is warmed up in August. Below 30 m, the difference between profiles is less. Since the sound velocity profiles do not differ significantly at the source depth, the damping is comparable for the two months and the received level at different distances similar to a pile strike with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s (Figure 4, left). If the source depth was shallower, there would be a difference between the months as with the Kattegat. This sound propagation can be considered rather extreme – but not rare – in the area. The two vertical lines indicate two of the recommended harmful sound exposure levels for fish and harbour porpoises, respectively (Figure 4, right). If the source level changes, the distances will also change. Table 4 shows the received sound exposure levels at different distances when the equivalent source level was set to $SEL_{(ss)}$ 226 dB re 1 µPa²s. Here, the similarities in damping as a function of distance are clear.

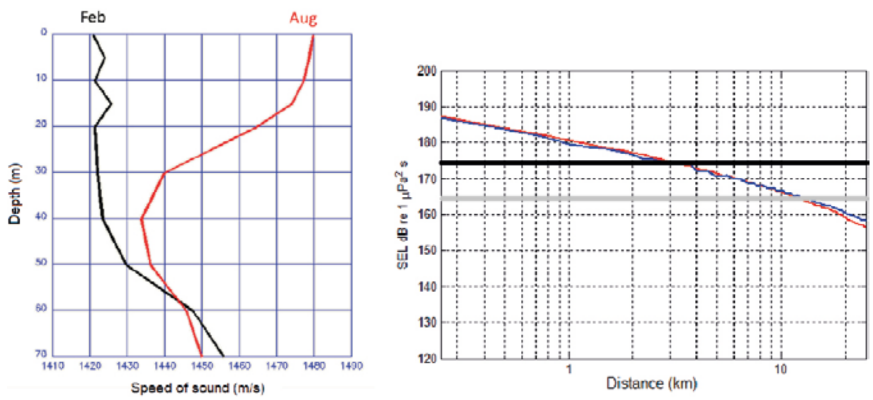



Figure 4. Left: Sound velocity profiles for February (black) and August (red) in Hanöbukten. Sound velocity profiles for February (black) and August (red). Right: The sound propagation for a pile strike with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s as a function of distance along the bearing of 90° to 270° in Hanöbukten. The calculations were made with sound velocity profiles typical for February (blue) and August (red). The grey line shows the recommendation on harmful noise levels for TTS for harbour porpoises ($SEL_{(ss)}$ 164 dB re 1 µPa²s), and the black line mortality and injury to the internal organs of fish ($SEL_{(ss)}$ 174 dB re 1 µPa²s). Note that the injury also depends on the number of sound pulses over time. The figure shows examples of modeled sound propagation with special sound propagation conditions and assumptions about exposure (see explanation in Section 2.5).

**Table 4. Sound exposure level as a function of distance in Hanöbukten when the equivalent source level was set at $SEL_{(ss)}$ 226 dB re 1 µPa²s.**

| Distance from source (km) | $SEL_{(ss)}$ (dB re 1 µPa²s) FEB. | $SEL_{(ss)}$ (dB re 1 µPa²s) AUG. |
|---|---|---|
| 0.75 | 182 | 182 |
| 1.5 | 178 | 179 |
| 3 | 175 | 175 |
| 5 | 171 | 171 |
| 10 | 167 | 166 |
| 20 | 160 | 159 |

### 2.5.3 The Southern Baltic

In the Southern Baltic Sea in the vicinity of the Midsjö banks, it is assumed that the bottom consists of a 20 m thick sand layer on top of crystalline bedrock. The salinity was set at 7.8 PSU. The calculations were made out to a distance of 25 km. The water depth in the area along a bearing of 90° to 270° ranged between 20 and 50 m. The sound velocity profiles indicate large differences down to 40 m as the surface water is warm in February and then evens out at greater depths (Figure 5, left). In August, a downward-refracting profile is apparent, resulting in greater mitigation of the noise than for February. As a result, the noise becomes louder at a distance greater than about 2 km in February compared with August (Figure 5, right). For these calculations, an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 μPa²s was used. This sound propagation can be considered rather extreme – but not rare – in the area. The two vertical lines indicate two of the recommended harmful noise exposure levels for fish and harbour porpoises, respectively. If the source level changes, the distances will also change. Table 5 shows the received sound exposure levels at different distances when the equivalent source level was set to $SEL_{(ss)}$ 226 dB re 1 μPa²s.

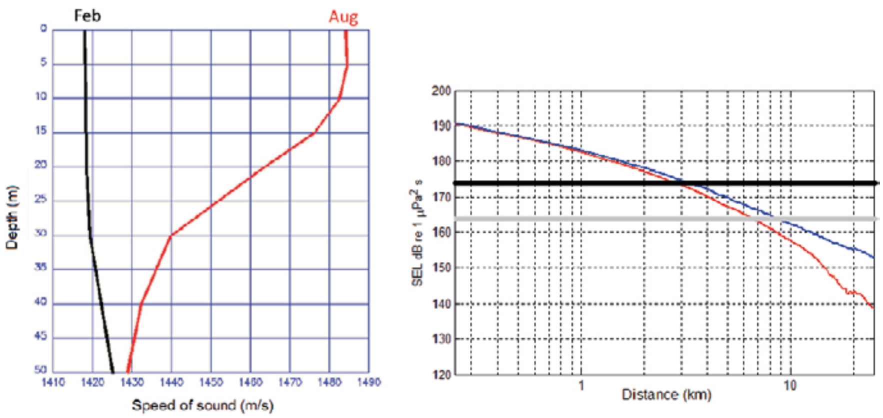



Figure 5. Left: Sound velocity profiles for February (black) and August (red) in the Southern Baltic Sea. Right: The sound propagation for a pile strike with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 μPa²s as a function of distance along the bearing of 90° to 270° in the Southern Baltic Sea. The calculations were made with sound velocity profiles typical for February (blue) and August (red). The grey line shows the proposed harmful noise level for TTS in harbour porpoises ($SEL_{(ss)}$ 164 dB re 1 μPa²s), and the black line mortality and injury to the internal organs of fish ($SEL_{(ss)}$ 174 dB re 1 μPa²s). Note that the injury also depends on the number of sound pulses over time. The figure shows examples of modeled sound propagation with special sound propagation conditions and assumptions about exposure (see explanation in Section 2.5).

**Table 5. Sound exposure level as a function of distance in the Southern Baltic Sea when the equivalent source level was set at $SEL_{(ss)}$ 226 dB re 1 μPa²s.**

| Distance from source (km) | $SEL_{(ss)}$ (dB re 1 μPa²s) FEB. | $SEL_{(ss)}$ (dB re 1 μPa²s) AUG. |
|---|---|---|
| 0.75 | 185 | 184 |
| 1.5 | 180 | 180 |
| 3 | 175 | 173 |
| 5 | 170 | 167 |
| 10 | 162 | 159 |
| 20 | 155 | 143 |

### 2.5.4 The Bothnian Sea

In the Bothnian Sea, the bottom is assumed to consist of a 5 m thick layer of clay on top of a 20 m sand layer underlain by bedrock. The salinity was set at 6.5 PSU. The calculations were made out to a distance of 25 km. The water depth in the area along the bearing of 90° to 270° ranged between 70 and 90 m. The source depths were chosen to give rise to the highest possible levels, which for February is 35 m and for August 52 m. At these depths both of the sound velocity profiles are nearly the same, which explains the lack of any major difference in noise levels for February and August (Figure 6, left). This means that the received noise level at different distances will be the same regardless of month (Figure 6, right). For these calculations, an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s was used. This sound propagation can be considered rather extreme – but not rare – in the area. The two vertical lines indicate two of the recommended harmful noise exposure levels for fish and harbour porpoises, respectively. If the source level changes, the distances will also change. Table 6 shows the received sound exposure levels at different distances when the equivalent source level was set to $SEL_{(ss)}$ 226 dB re 1 µPa²s. Here, the similarities in damping as a function of distance are clear.

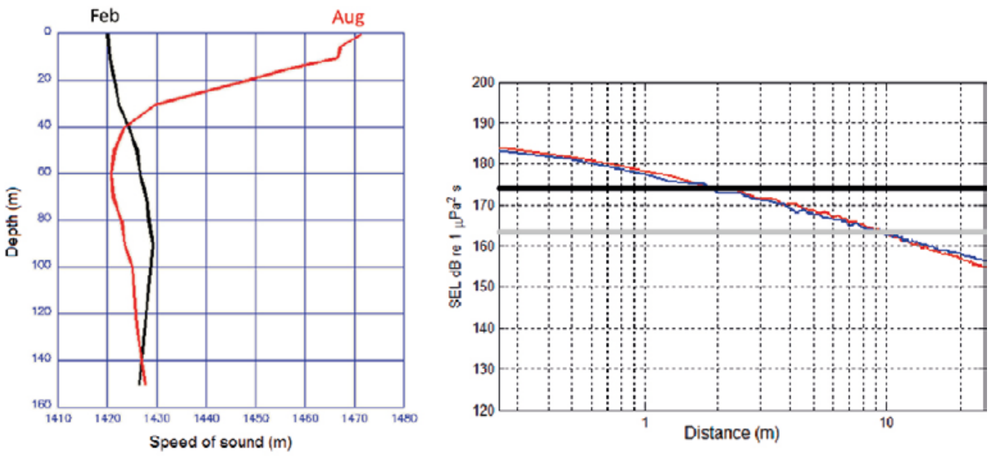



Figure 6, Left: Sound velocity profiles for February (black) and August (red) in the Bothnian Sea. Right: The sound propagation for a pile strike with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s as a function of distance along the bearing of 90° to 270° in the Bothnian Sea. The calculations were made with sound velocity profiles typical for February (blue) and August (red). The grey line shows the recommended threshold for TTS for harbour porpoises ($SEL_{(ss)}$ 164 dB re 1 µPa²s), and the black line mortality and injury to the internal organs of fish ($SEL_{(ss)}$ 174 dB re 1 µPa²s). Note that the injury also depends on the number of sound pulses over time. The figure shows examples of modeled sound propagation with special sound propagation conditions and assumptions about exposure (see explanation in Section 2.5).

**Table 6. Sound exposure level ($SEL_{(ss)}$) as a function of distance in the Bothnian Sea when the equivalent source level was set at 226 dB re 1 µPa²s $SEL_{(ss)}$.**

| Distance from source (km) | $SEL_{(ss)}$ (dB re 1 µPa²s) FEB. | $SEL_{(ss)}$ (dB re 1 µPa²s) AUG. |
|---|---|---|
| 0.75 | 178 | 179 |
| 1.5 | 174 | 176 |
| 3 | 171 | 171 |
| 5 | 168 | 169 |
| 10 | 162 | 163 |
| 20 | 159 | 157 |

# 3 Pile driving as a sound source

## 3.1 Installation techniques

To stabilise the ground prior to building houses, railways or bridges, or as the foundation for offshore wind turbines, the construction industry uses pile driving. Different techniques are currently available for driving the piles into the ground. The main techniques include impact pile driving, vibratory pile driving and drilling. In many cases, a combination of these techniques is used to drive the pile to the desired depth. The piles can vary in diameter, from a few decimetres for port structures up to six to eight metres for an offshore wind turbine monopile (Figure 7). To anchor a jacket foundation or tripod to the seabed, smaller piles are also used that are driven into the seabed, and in some cases the pile driving takes place completely under water. In cases where a specific area of land must be reinforced, piles of metal, wood or concrete can also be used. An undesired effect of these installation techniques is the high noise levels they produce, which adversely affect marine organisms (Andersson and Sigray, 2009; OSPAR, 2009). This probably occurs even when the pile driving takes place on land in the vicinity of the water and not in the water itself, since vibrations travel down into the ground and out into the water. However, there is a lack of measurements and studies on the severity of pile driving's effects in the water during this cirumstance. Many factors come into play in this scenario, such as the composition of the seabed and soil, the pile driving technique and the distance to the water. The following sections briefly describe some of the main installation techniques in use today for offshore wind power. (Some examples and data are from land-based operations.) Many more installation techniques exist, but they are either in the trial stage or have not been used to any great extent. For a more detailed description on this topic, see the reports by Saleem (2011) and OSPAR (2014).

Figure 7. Left: Pile driving of a port structure in Ålseund, Norway (Photo: Mathias Andersson, FOI). Right: A pile driving ship in the German North Sea prepares the installation of an offshore wind turbine foundation (Photo: Markus Linné, FOI).

### 3.1.1 Impact pile driving

During impact pile driving, a hydraulic or diesel hammer strikes a pile, pipe or beam into the ground or the seabed. Falling weights are also used to drive a pile into the ground. The hammer impact energy level or the weight of the falling weight can vary greatly depending on the type of pile and soil conditions. Basically, more energy and heavier weights mean more noise emitted. There are some hammers that can strike at up to 3,000 kJ, and these are used today in the offshore industry. In the future, the largest piles/foundations (monopiles) for offshore structures are expected to be able to reach up to 8 m or more in diameter. The monopile foundation for offshore wind power works best for a water depth of 10–35 m, and about 80% of all wind turbines today rest on driven monopile foundations (OSPAR, 2014). In order to build at greater depths, even larger foundations are needed. According to Saleem (2011), foundations of at least 7–8 m in diameter are needed for 40 m depths; these remain in the demo phase for such depths. Jacket or tripod foundations can be placed in deeper waters (down to 50–60 m) and, in these cases, the legs are installed attatched to the seabed using smaller piles than traditional monopiles (Hammar et al., 2008; Saleem, 2011). In Sweden, a few wind farms have been built using impact pile driving, like the Utgrunden in Kalmarsund (McKenzie-Maxon, 2000). According to Saleem (2011), impact pile driving brings the advantages of a simple design, proven technology and an ability to handle many different soil conditions. In addition, it is relatively easy to calculate the bearing capacity for the monopile in different soil conditions, that is, how well the sediment holds the pile in place. One disadvantage of this method is that it generates the highest noise levels. Also, steel is expensive. So the deeper you need to install the more steel you need, and the heavier the structure becomes. And when it's time to decommission the foundation, it is not possible to remove the entire structure – whatever is attached to the seabed or ground will remain.

### 3.1.2 Vibratory pile driving

Another method used is vibratory pile driving, or vibropiling. With this method, piles are vibrated into the ground at a frequency of about 20–40 Hz (OSPAR, 2014). Rotating counterweights induce this vibration and facilitate penetration of the bottom. For large piles, multiple vibrating systems can be used. Sometimes it can be necessary to impact pile drive a particular portion due to the varied seabed structure (e.g., hard structures such as boulders) during vibratory pile driving. Because the vibrations are relatively low-frequency, the so-called cut-off frequency can mean that it can be mitigated by the bottom in certain frequencies. This only applies for shallow water depths lower than 40 metres or so. The combination of vibratory and impact pile driving allows the total noise level to be lower than it would be using impact pile driving only, because a fewer number of strikes are needed to drive the pile into the bottom. In some cases, a lower average broadband level of 15 to 20 dB was measured compared with regular impact pile driving (Elmer et al., 2007a; Betke and Matuschek, 2010). However, it exhibits a more continuous nature, which makes the comparison of noise levels from impact pile driving – which exhibits a more impulsive nature – not entirely accurate (CSA Ocean Sciences Inc., 2014). At present, a few offshore wind farms have been successfully installed in the U.K. and the Netherlands using vibratory pile driving, in whole or in part (OSPAR, 2014). In these cases, foundations with a diameter of up to six metres have been used. The suitability of either vibratory pile driving or impact pile driving depends mainly on the composition of the seabed or ground where pile driving is to take place. The advantages of vibratory pile driving, according to Saleem (2011), are the following: there are no size limitations for the foundation; it is often faster than impact pile driving; it is cheaper; it creates generally lower levels of radiated noise; and the vibration technique can be used when decommissioning a foundation. The disadvantages are that the strength is difficult to estimate, the construction is less reliable and, oftentimes, some sections must be impact driven anyway. In addition, the technique involves a more complex handling of the necessary equipment.

### 3.1.3 Drilling

A quieter but more complicated installation method is drilling, and as with vibratory pile driving the noise is more continuous compared with impact pile driving. Much experience has been gained from drilling in the offshore industry. In broad terms, drilling involves inserting a drill inside the foundation and lowering it down to the bottom. The drill is anchored in the bottom, and residual material is transported up through the foundation. In some cases, the pile can be driven into the bottom at the very end to anchor it better. The nature of the bottom determines whether drilling is possible. It is mainly suitable when harder materials in the bedrock are present, such as limestone or sandstone, but also for sandy bottoms that contain large rocks. Monopile foundations up to a diameter of 4.5 m have been drilled in the U.K. (OSPAR, 2014). One of the first Swedish offshore wind farms, Bockstigen (near the southern tip of the island of Gotland) was installed using drilling. The advantages of drilling are that you can install monopiles in very hard seabeds or bottoms with different strengths and the noise level is lower than with impact pile driving. The disadvantages are that drilling takes longer than impact pile driving and creates a lot of material requiring removal. Also, you often have to drive the pile at the very end, and the bearing capacity is difficult to calculate.

### 3.1.4 High frequency, low energy pile driving

A variant of impact pile driving called HiLo uses a technique whereby the impact energy level decreases but the strike rate increases. Normal impact pile driving has a strike rate of about 40 strikes/minute, but the HiLo technique uses about 90 strikes/minute. The decreased impact energy also reduces the radiated noise. This is not currently a common practice but it has been tested in Germany (Wilke et al., 2012).

## 3.2 Pile driving as a sound source

To understand and estimate the noise associated with pile driving, you must have a good source model and a reasonable propagation model. The modelling should be divided into near field and far field. If you perform measurements at a certain distance, you will want to use an appropriate model for calculating and predicting the noise field at other distances. You also need to make some assumptions. These can be:

- The pile driving takes place in shallow waters at a depth between 20 and 100 metres.
- The noise level in the far field has a simple distance dependence that can be described as a linear relationship between sound pressure level (SPL) in dB and distance. But in the near field, the relationship is complicated and cannot be described as a linear relationship between SPL dB and distance.
- The bottom always consists of sediment. The sediment's properties are obtained through a ground survey prior to the installation phase. Soft sediment does not require the same hammer energy as harder and firmer sediment.

### 3.2.1 Sources

Pile driving as a noise source is described by de Jong and Ainslie (2008) as a mechanical mass (hammer or rammer) that hits the pile in a direction vertical to a small surface area equal to the cross section. The mass can weigh up to 1,000 kg. The speed upon impact with the pile is around 10 m/s. Mainly, longitudinal waves (P-waves, i.e. primary waves) are generated, which have the highest speed in solid materials, i.e., they reach a certain point before subsequent wave types (phases) arrive. When calculating the total energy, however, both the P-wave pulse and other phases might need to be considered. P-waves generate a radial displacement, resulting in the horizontal portion of the sound field. This becomes the source of noise out in the water column. Closest to the pile in the near field, the propagation is non-linear and there are no empirical models for this region (Figure 8). The border between the far field and near field (D) can be estimated by equation (1), where A is the pile dimension and λ is the wavelength. One can see that there is a risk of overestimating the source level if pure extrapolation is utilised. A pile 80 m long (speed of sound 1,500 m/s and at 10 Hz) will be able to have a near field region up to about 150 m.

$$D = A^2/\lambda \qquad \text{eq (1)}$$

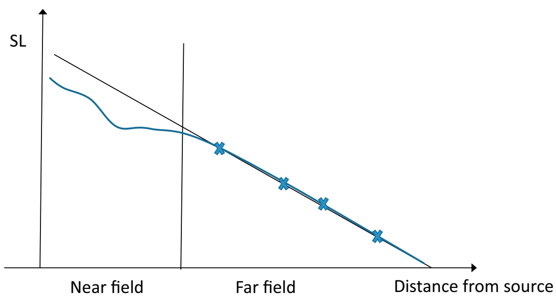



Figure 8. Illustration of the sound field's near field and far field during pile driving, modified from Nedwell and Howell (2004).

The following key variables that affect the generated noise level at the source have been identified by de Haan et al. (2007):

- Hammer impact energy. If you incrementally increase the impact energy, it will reach its intended level after a certain predetermined period of time. This energy increase should be adapted to the piling area's geological bottom conditions, water depth and environmental protection requirements. Since the total energy is linked to the repetition frequency of the strikes, which can vary from less than 1 second up to several seconds, it also provides an input value for planning.
- The material, diameter and length of the pile.
- How deep in the sediment the piles are to be driven.
- The bottom or ground composition and the pile driving resistance.

Because a radial pressure component occurs down the length of the pile, the source can be described as a line array source. This results in the persistence of the horizontal line source even when the pile has begun to penetrate the sediment. Beyond the near field, one can assume that the source has the characteristics of a point source.

You can divide the radiated pile-generated noise field at the source in three sections starting from the near field (Figure 9); see also Massarsch and Fallenius (2008).

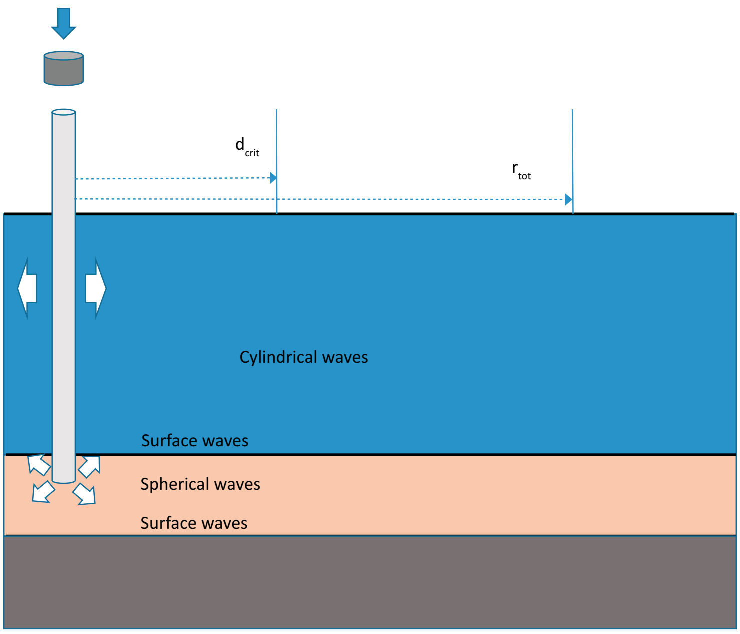



Figure 9. Soures of noise associated with pile driving, modified from Massarsch and Fellenius (2008).

These sections consist of surface waves at the ground surfaces, cylindrical waves propagating laterally from the pile shaft, and spherical waves that emanate from the pile down into the bottom. These three components coincide at a distance $r_{tot}$. The distance $d_{crit}$ in Figure 9 is the minimum distance from the pile where the pressure waves from the pile toe and the waves – radially outward from the pile as well as from the hammer at the top – can together converge at the surface. Another study that also describes the characteristics of the source both theoretically and experimentally is the Reinhall and Dahl study (2011). They arrive at a similar result: that the energy producing the noise field in the water consists mainly of radial waves from the pile. They additionally conclude that there are three main source components: a radial displacement wave from the pile, a cone out from the pile's toe in the sediment, and a cone out from the top of the pile where the hammer strikes the pile.

The bottom conditions and driving resistance affect the amount of noise energy generated in the seabed and the water during pile driving. Figure 10 illustrates two different bottoms, one with sandy soil (Case 1) and one with soft soil on a stiffer bottom layer (Case 2). During pile driving in Case 1, the radiated noise will increase almost linearly with increased driving depth and resistance. In Case 2, on the other hand, we see a slightly elevated level when the driving begins. This is because the process is rather easy in the beginning with a relatively low level of radiated noise until the pile reaches the harder bottom layer where the resistance is higher. The resistance of the bottom to the pile is the single largest source of noise generation during pile driving. This is true whether on land or at sea. Ground movements are not directly linked to impact energy but are rather a combination of several factors. In addition to resistance in the bottom sediment and the impact energy, the radiated noise energy is affected by pile length, diameter and material. Most of the energy travelling through the pile continues out into the bottom. The part of the pile located above the bottom in the water directly generates only about 1% of the energy directly into the water as acoustic energy. But this small portion of the driving energy can, however, attain very high pressure levels in the water. The remaining energy is transmitted down the length of the pile, and out into the water column and into the bottom (Figure 11). This energy also contributes to the pile driving noise that is generated. Much of the energy that is transmitted into the bottom can be converted into other forms of energy such as heat; see Elmer et al. (2007b). The noise can, of course, also be converted from the bottom to the water, especially if there is a stiff bottom. The high-frequency components of the noise from the bottom to water column are mitigated more than the low-frequency components. The range (or the lower propagation losses) of the low-frequency noise components in the bottom sediment, however, is much larger. Since the bottom, pile, water and air interact when noise is generated, it is important to take all the components into consideration at the source, especially when choosing noise-mitigation methods. Simply studying the horizontal distance at short distances between the noise source (pile location) and the measurement site might lead to an incorrect ratio between estimated and actual noise level.

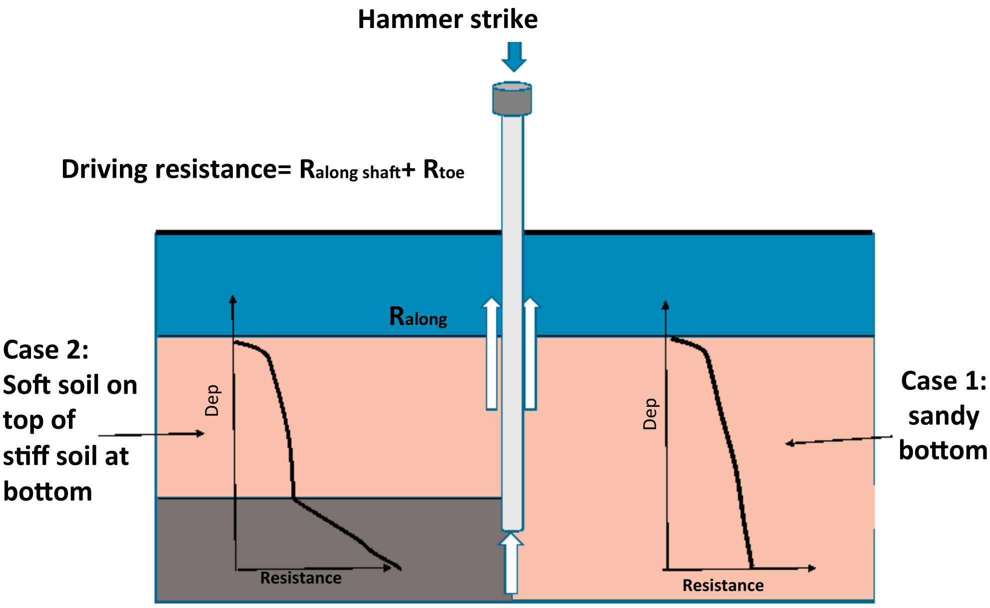



Figure 10. Conceptual image of how the bottom's driving resistance (R = resistance) is affected by the bottom type for two different layers: sandy (Case 1) and soft soil on top of a stiff bottom layer (Case 2). Modified image from Massarsch and Fellenius (2008).

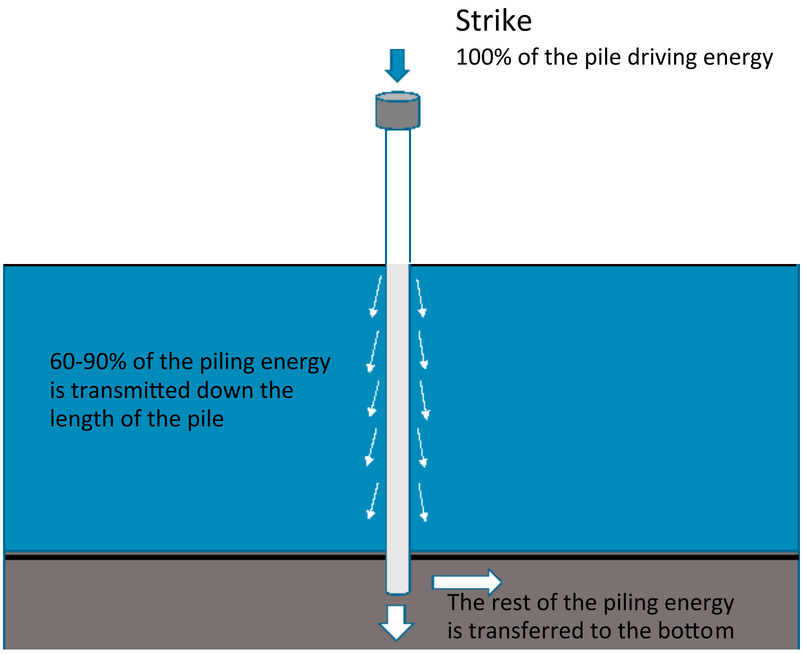



Figure 11. Proportion of radiated energy from the piling hammer in the different sections, bottom and water column in relation to total energy. Only a fraction of the energy is transferred directly into the water. Most of it is discharged only after transmission in the pile. Revised figure from Elmer et al. (2007b).

### 3.2.2 Factors that affect noise radiation

When pile driving in sediment with the presence of loose deposits, there are four major factors that affect the radiating noise generated by impact pile driving (Elmer et al., 2007b). These are:

1) Wave propagation in the pile: the energy generated by the blow of the hammer against the pile and transmitted through the pile
2) The transition between pile and ground: down the length of the pile to the bottom of the pile
3) Wave propagation in the water, the ground and the seabed: acoustic impedance and properties in and between the different media
4) Dynamic properties: the dynamic response in the pile, bottom substrate and all structures. These describe how the material transports impact energy without becoming deformed.

When a pile is driven slowly into the ground or seabed, the resulting vibrations or noises are very weak or non-existent. When the penetration speed increases, so does the noise intensity. There is a dependency between the dynamic forces at play in the interface between the pile and the surrounding media. The noise is caused by the shock wave created in the pile when the hammer strikes it. This shock wave propagates through the pile and is transmitted along the pile shaft and at the toe to the bottom which the pile penetrates. The noise that is transmitted from the shaft and toe to the surroundings is different wave types, for example pressure waves. The pile driving noise often decreases as the distance increases from the vibration source but may be strengthened in the bottom layers or structures due to resonance effects. What affects the source level at a given location is therefore an integration effect between the elasto-dynamic characteristics of the bottom substrate, pile material, pile dimensions, hammer material, impact energy, stroke rate, water depth and the pile's penetration depth.

### 3.2.3 Metrics and units

It is important to define different metrics in order to quantify and describe the noise levels in the ways that are necessary from an environmental point of view. Unlike for noise in air acoustics, very few standards are available for underwater noise. In the United States, the American National Standards Institute (ANSI) has published a series of standards for measuring noise from shipping vessels as well as certain terminology standards. But at the ISO level these efforts have just begun; see Section 3.4 on sound emission standards.

Pile driving with hammer strikes is a noise source that can be regarded as an impulsive sound (non-stationary noise). "Impulsive" refers to a discrete sound event of short duration that distinguishes itself from other noise; see Vaseghi (2000). It is important to characterise noise as either impulsive or non-impulsive because its effects differ significantly depending on which auditory organ you compare with (Southall et al., and 2008). The definitions of short duration and discrete depend on the area of application.

An impulse is defined as a change in sound pressure with a rapid rise-time from ambient pressure to a maximum pressure value, followed by a rapid decrease (Southall et al., 2007). Within acoustics, impulsive sound is often defined as one or more pulses which have a duration of at least 1 s (see ISO 10843:1997). This is related to human hearing. In underwater acoustics, an impulse with a duration less than 0.2 s is designated as short and impulsive. Furthermore, Southall et al. (2007) divide impulsive sound (impulse) into two categories: slow (1 s) and fast (0.125 s). An impulse can, at a specific distance and in a particular environment, also be considered to be a signal that does not meet the criteria for a pulse and is converted to a non-impulse. In a different environment and surroundings, the same signal can be preserved as an impulse.

Different metrics are necessary in order to describe the intensity and energy of a driving pulse (see illustration, Figure 12). The literature contains many different names for the same metric. This study mainly uses designations contained in the draft ISO standard ISO/TC 43/SC 3 Underwater acoustics. Below, *L* is used to denote the sound pressure level. The most useful definitions related to pile driving noise are:

1. Peak-to-peak sound pressure level $SPL_{(peak\text{-}peak)}$ in dB re 1 µPa is the difference between the maximum and minimum overpressure and underpressure according to $L_{(peak\text{-}peak)} = 20\log_{10}(\max(p(t)\text{-}\min(p(t))$, where *t* means that the measure is calculated in the time series.
2. Peak sound pressure level $SPL_{(peak)}$ in dB re 1 µPa is the maximum absolute value of the overpressure or underpressure according to $L_{(peak)} = 20\log_{10}(\max(|p(t)|)$.
3. Root mean square (RMS) sound pressure level in dB re 1 µPa. For a single pulse or a series of pulses. Is a measure of the pulse energy $L_{(peak)} = 20\log_{10}(\mathrm{rms}(p(t))$. Often totalled over 1 s of the time series or to a moving average. In the latter case, it is important to state the window over which RMS was estimated, i.e., when you start and end the calculation.
4. Sound exposure level (SEL) is the sound that you are exposed to. SEL is defined as the cumulative constant sound pressure level in a window that is long enough to accommodate an entire single pulse $SEL_{(ss)}$ and that has the same energy as the reference in the corresponding 1 s window. This measure is calculated in a time window from T1 to T2 (Figure 12). T1 begins when the signal level exceeds 5% of the background, and T2 when the level has gone down corresponding to 95%. Sound exposure (SE) is a measure of the acoustic energy expressed in the unit decibel re 1 micropascal squared-second (dB re 1 $\mu Pa^2 s$). SEL allows the comparison of radiated noise with different window lengths and content. If there are several pulses (driving a foundation can take up to two hours), then we can summarise these as a cumulative measure of acoustic energy $SEL_{(cum)} = SEL_{(ss)} + 10 \log(n)$, where *n* is the number of pulses/strike. It is a good idea to then state the time duration and number of pulses that the cumulative value applies to.

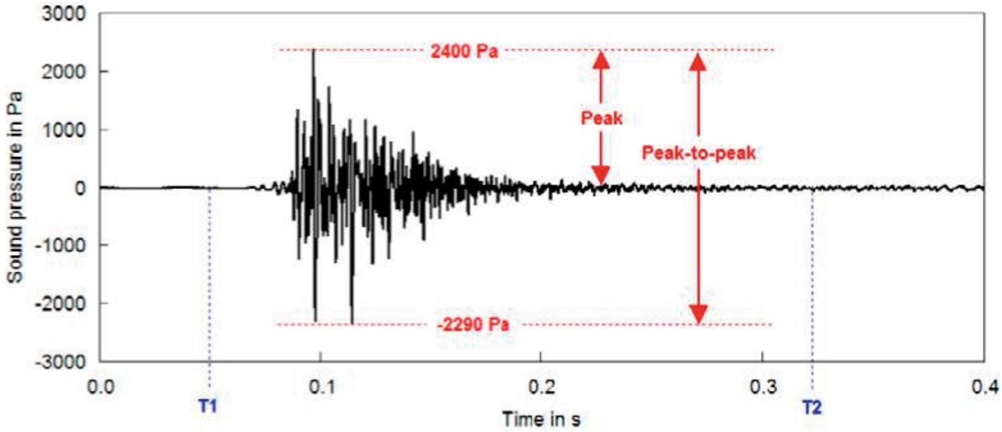



Figure 12. Example of $SPL_{(peak-peak)}$ value and $SPL_{(peak)}$. They are calculated in a time window from T1 to T2. T1 begins when the signal level exceeds 5% of the background, and T2 when the level has gone down corresponding to 95%; from Betke (2008).

In the de Jong and Ainslie study (2008), the authors highlight the importance of a good correspondence between model and metric. An example of a source signal of pile driving from an offshore wind farm at a distance of 720 m and a hammer energy of 850 kJ can be seen in Figure 13. One can see that the noise is made up of two components, a harmonious first component that is more low-frequency (i.e., there is a greater distance between the peaks) which is then followed by a more high-frequency package (shorter distance between the peaks). These have then propagated along different paths with different mitigation through the water and bottom.

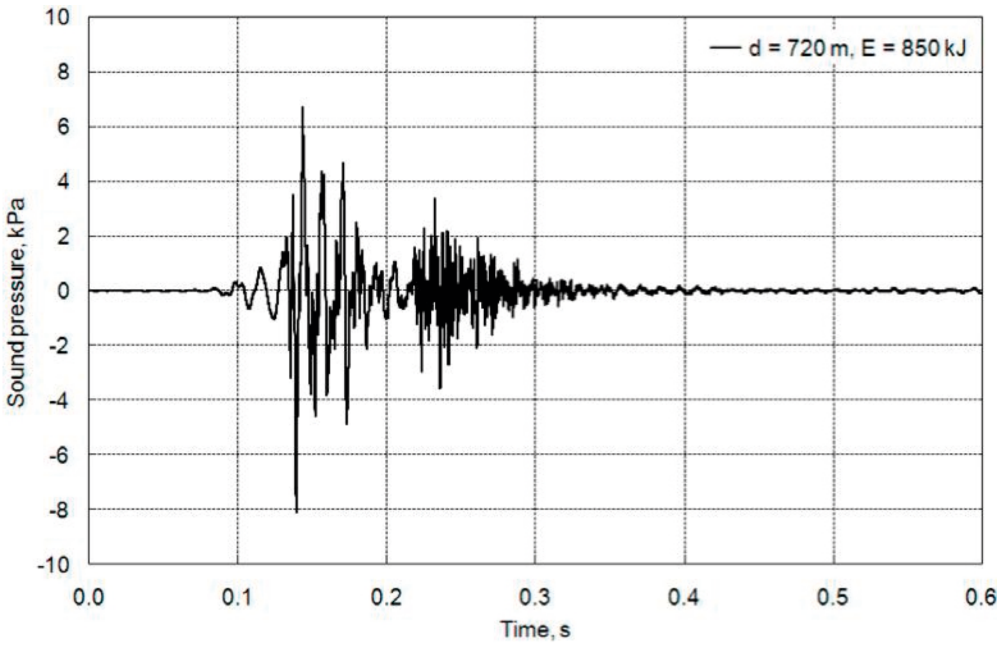



Figure 13. Typical registered signal for a single pile strike where a more low-frequency component is shown first, followed by a more high-frequency component; from Betke (2008).

Common practice is that if you have a peak value then you can add the 6 dB to obtain the peak-to-peak value. RMS refers to the average sound pressure over a given unit of time. 1 s is a common average measurement for acoustics. If the pulse length is shorter than 1 s, SEL should be corrected using SEL = SPL + 10 log10 T, where *T* is the pulse duration in seconds, for example 0.3 s. Cumulative SEL increases the level by 10 dB for each 10-fold increase in the number of pile strikes. The equivalent continuous sound level, $L_{eq}$, is also often used. It is the RMS value over the entire measurement period, even the data between the pulses. The relationships between the different metrics are highlighted in Figure 14.

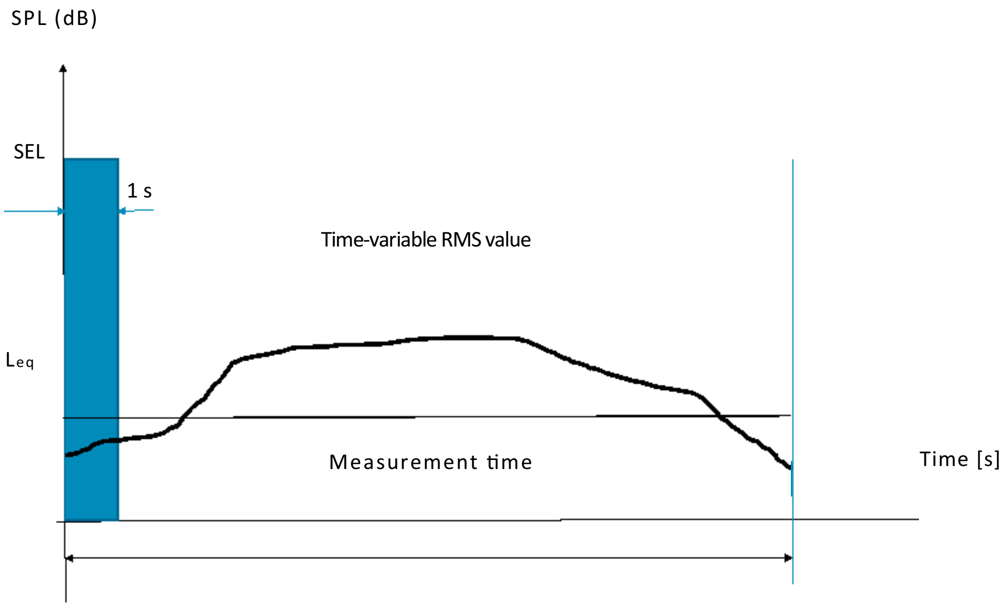



Figure 14. Illustration of relationship between SPL, RMS, SEL and Leq.

SEL can then be calculated for a specific location and time. The level should be stated in dB re 1 µPa²s. It is calculated over a single pulse for a specific time (T1 to T2 in Figure 12) or as a total of all pulses cumulatively. For multiple pulses, these are totalled per pulse as:

$$L_{tot} = 10\log_{10}\left(\sum_{k=1}^{n} 10^{L_k/10}\right)$$

where Lk is the sound pressure for each pulse. Since the SEL values can have significant dynamics, an upper and lower level are used between which the SEL value should be located, e.g., $SEL_{05}$ to $SEL_{90}$. $SEL_{05}$ means that the SEL values exceed the upper 5% of the data, and $SEL_{90}$ means that the values exceed the level by 90%. $SEL_{50}$ is the same as the mean. This mathematical exercise is illustrated in Figure 15. Depending on which of these levels you use, the estimated total (over all frequencies) SEL value is different. It is important that future recommendations are clear about which level is referred to. Furthermore, a wideband SEL does not take into account animal hearing sensitivity. More on this in Chapter 5.

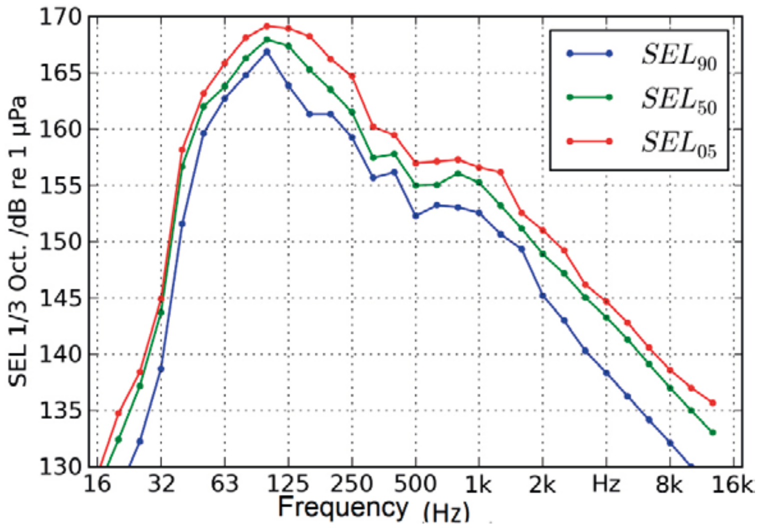



Figure 15. Third-octave band spectrum of sound exposure level (SEL) from a pile driving series and the proportion of SEL; $SEL_{90}$ is when 90% of the level is exceeded, $SEL_{50}$ is when half is exceeded, and $SEL_{05}$ is the 5% highest levels. The figure is from ITAP in Germany and shows the data used for the modelling examples.

### 3.2.4 Measured sound levels

Pile driving as a sound source generates broadband noise but with most of the energy between 100–1,000 Hz (see example in Nedwell and Howell, 2004; de Haan et al., 2007; de Jong and Ainslie, 2008; Bailey et al., 2010; Norro et al., 2013). Many more measurements have been made than those published in scientific articles or reports, but the technical reports from measurement programmes are often inaccessible due to business confidentiality. Table 7 presents a summary of several noise levels that were measured when driving different foundations. The levels differ quite a bit as the bottom type and water parameters affect the outcome, but you can still note a trend of increased noise level as a function of pile diameter. This relationship is also shown in Figure 16.

As previously described, several factors contribute to the radiated noise, such as the pile's length, diameter and material. In recent years, Germany has built a number of wind farms, and their control programme has almost always included measuring the driving noise during the design phase without mitigation measures. Bellmann (2015) indicates noise levels (± 5 dB) in SEL from a great many measurements during construction work, with piles having a diameter of 0.8–6 m (Figure 16). It is clear that the noise level increases as diameter increases, and it is likely that pile diameters will increase in the future in order to build in deeper waters. The exponentially curve-fit noise level in Figure 16 shows that the level at 750 m could rise to above SEL 180 dB re 1 µPa$^2$s and above $SPL_{(peak)}$ 205 re 1 µPa for a pile with a diameter of 8 m in the absence of mitigation measures.

**Table 7. A comparison of noise emitted during construction of various windpower structures without mitigation measures, sorted by pile diameter. $SPL_{(peak)}$ refers to either the measured level at 750 m or normalised to 750 m as well as expected $SEL_{(ss)}$ where possible. Data from Betke 2008; de Jong and Ainslie 2008; Norro et al. 2013; Kosecka et al. 2015; OSC 2015; Yang et al. 2015.**

| Site | Year | Pile diameter (m) | $SEL_{(ss)}$ (dB re 1 μPa²s) | $SPL_{(peak)}$ (dB re 1 μPa) |
|---|---|---|---|---|
| Port construction | 2005 | 0.9 | 157 | 183 |
| Port construction | 2005 | 1 | 159 | 185 |
| Fino 1 | 2003 | 1.6 | 162 | 184 |
| C-Power, phases 2&3 | 2011 | 1.8 | 178 | 189 |
| Hong Kong–Zhuhai–Macau Bridge | 2014 | 2 | 167 | 191 |
| Alpha Ventus | 2008 | 2.7 | 174 | 199 |
| Utgrunden | 2000 | 3 | 166 | n/a |
| Sky 2000 | 2002 | 3 | 163 | 189 |
| Fino 2 | 2006 | 3.3 | 169 | 189 |
| Amrumbank West | 2005 | 3.5 | 171 | 191 |
| Horns Rev II | 2008 | 3.9 | 176 | 195 |
| North Hoyle | 2003 | 4 | n/a | 194 |
| Q7 | 2007 | 4 | 177 | 200 |
| Barrow | 2005 | 4.7 | n/a | 195 |
| Fino | 2008 | 4.7 | 172 | 196 |
| Belwind | 2010 | 5 | 166 | 194 |
| Northwind | 2013 | 5 | n/a | 196 |
| Kentish Flats | 2015 | 5 | 180 | n/a |

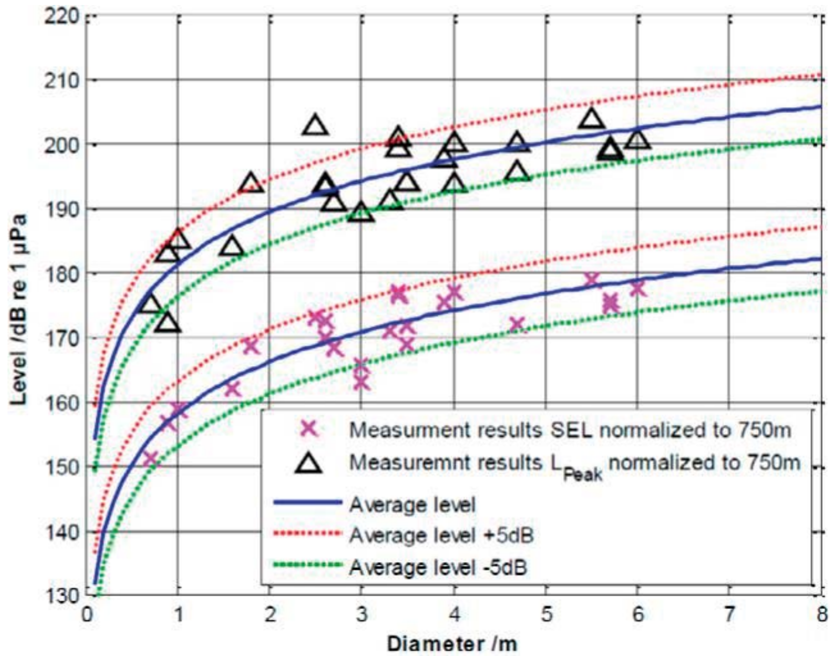



Figure 16. Measures noise levels from underwater piling in the form of $L_{peak}$ = $SPL_{(peak)}$ values and SEL values normalised to 750 m as a function of pile diameter, and an exponential curve fitting to the data with a statistical dispersion of ± 5 dB. Note that the Y axis is calibrated in dB re 1 μPa for Lpeak and dB re 1 μPa²s for $SEL_{50}$; from Bellman (2014).

The hammer impact energy will directly affect the radiated noise energy. Increased hammer energy leads to higher radiated energy (Figure 17) (Lepper et al., 2009). A simple rule of thumb is that by doubling the impact energy, the sound pressure level increases by 3 dB or through the function 10logE, where *E* is the relative increase in impact energy. This has been verified by measurements as in Betke (2008) and Bellmann (2014).

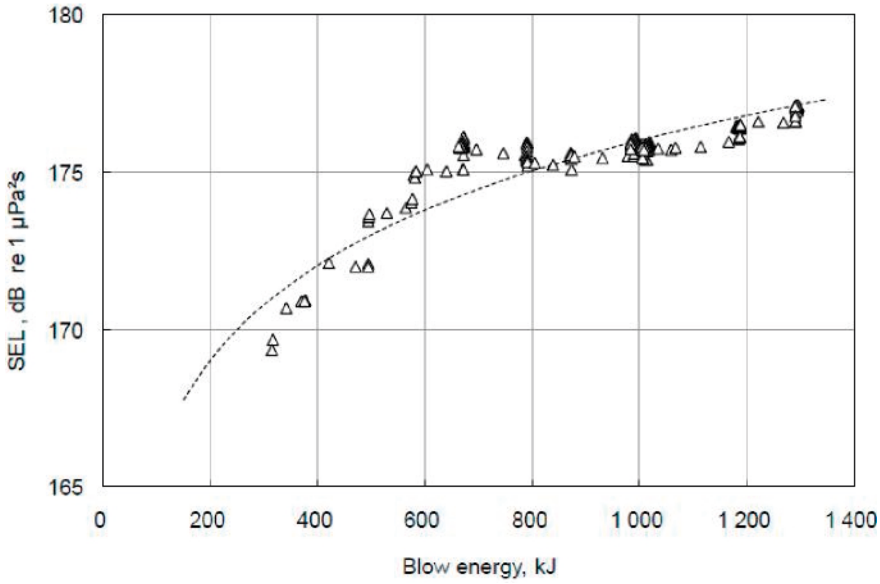



Figure 17. Sound exposure level (SEL) as a function of impact energy. Data is recorded at a distance of approximately 200 m from the pile driving activity. The depth is between 8–15 m; the dashed line is a logarithmic curve fitting; from Bellman (2014).

## 3.3 Frequency weighting

In order to consider the auditory sensitivity of a specific animal species or group, frequency weighting has been introduced. For marine mammals this is known as M-weighting (Southall et al., 2007; NOAA, 2015). This weighting deemphasises the impact from both high and low frequencies, as a band-pass filter similar to the A- and C-weightings for noise in aerial acoustics related to human hearing. There are many different M-weighting curves for different species and environments. An example of a weighting curve used for harbour porpoises in the North Sea and the Baltic is shown in Figure 18. Wideband noise that is not weighted will always give rise to an equally high or higher level than a weighted level. However, it is crucial that the correct weighting curve is used; if not, the risk of injury might increase (i.e., SEL might be underestimated). See Tougaard et al. (2015).

To simplify how the hearing organ functions and reacts to certain frequencies when exposed to broadband noise or sound, different bands have been developed to describe the ear's sensitivity relative to frequency and energy. The term *octave*, used in acoustics and music, corresponds to a doubling of frequency. In other words, there is one octave between 50 Hz and 100 Hz. One octave consists of three one-third octave bands. This means that each band is designated as a 1/3 octave band. For third-octave bands and octave bands, the bandwidth is proportional to a filter's geometric centre frequency. The centre frequency is used as a designation for each filter. In acoustic noise third-octave bands are often used, which are the same as one third of an octave band.

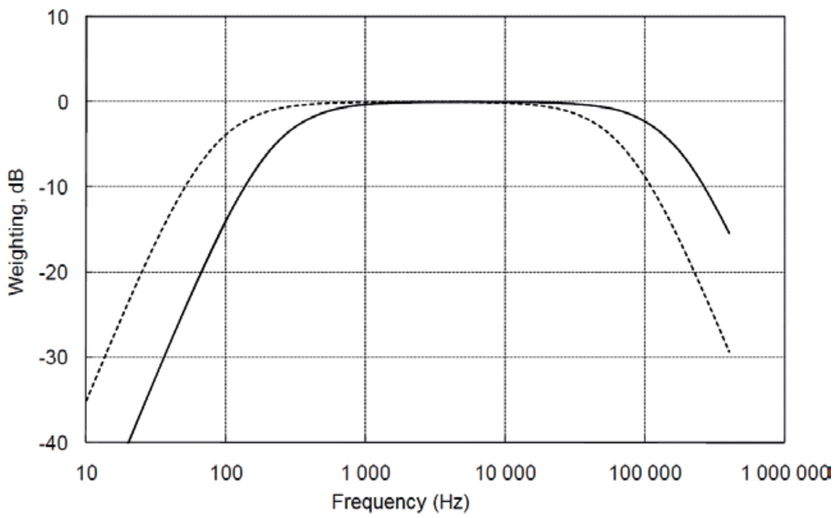



Figure 18. M-weighting curves for harbour porpoises (solid line) and eared seals (dashed line) from Betke (2008), modified after data from Southall et al. (2007).

## 3.4 Standards for acoustical measurements

An ISO working group is currently developing underwater acoustic terminology to clarify the meaning of the different units – SEL, SPL, source level, and so on – and how they are defined. An inofficial draft of the standard exists, but since several amendments are ongoing this publication cannot refer to them at present (according to a conversation with Pete Theobald, NPL). This working group is named "ISO/DIS 18405 Underwater acoustics – Terminology", and its work is expected to be completed in 2016–2017. In addition, there is a working group that is developing pile driving noise standards, "ISO 18406, on Underwater acoustics – Measurement of underwater sound radiated from percussive pile driving"[1]. A new working group is expected to begin work soon on ambient noise in the sea, but these efforts can take up to 3 years before being completed.

Because ISO standards have been lacking for how to measure pile driving noise and how to analyse and present data, several countries have published their own guidelines or recommendations. These are used to be able to make measurements in a standardised way, making the results comparable and

[1] Authors' comments: Since the first version of this report, the two ISO standards have been published and can be found on ISO's website.

serving as input when licensing offshore windpower operations. Below is a brief description of different countries' guidelines for measuring underwater noise and how they are related to pile driving licensing. Some countries have adopted thresholds in these documents for the impact on marine animals while others have separate guidelines. Chapter 6 contains a further description of the biological impacts.

**Germany** was early in publishing standards for windpower installations, and their guidelines contain both measurement methodology and documentation requirements (Müller and Zerbs, 2011, 2013; BSH, 2013). Their guidelines describe in great detail how to meet licensing requirements with regard to the thresholds for environmental impact.

**Denmark** is developing guidance for underwater noise during pile driving and the impact on marine mammals. A working group formed by energinet.dk, a company owned by the Danish Ministry of Climate, Energy and Building, has released the memorandum "Marine mammals and underwater noise in relation to pile driving" (Skjellerup et al., 2015) as well as a revison (Tougaard, 2015) made after several new field studies on the topic were published. However, this material contains relative little about measurement standards but rather focuses on the levels and thresholds that adversely affect marine animals.

In **Great Britain**, the National Physical Laboratory (NPL) has published a Good Practice Guide (NPL, 2014) for lack of guidelines from any responsible authorities. The guide was commissioned by Marine Scotland, The Crown Estate, and the National Measurement Office of the Department for Business, Innovation and Skills. However, the guide lacks a connection to the licensing of pile driving operations. Instead, this is regulated by the 1985 Food and Environmental Protection Act (FEPA) and Section 34 of the 1949 Coast Protection Act (CPA). The Joint Nature Conservation Committee (JNCC) has released a document describing the recommendations for preventing or minimising the environmental impact of pile driving noise (JNCC, 2010). It also provides recommendations for the formulation of a control programme.

For the **Netherlands**, the Netherlands Organisation for applied scientific research (TNO) has created standards that relate metrics, units and measurements (Ainslie, 2011; de Jong et al., 2011) as tasked by the Netherlands Minstry of Infrastructure and the Environment, Directorate-General for Water Affairs.

In **Ireland**, the Department of Art, Heritage and the Gaeltacht has published guidelines for managing and preventing the risk of disturbance from anthropogenic noise sources. The guidelines include pile driving as a noise source (Department of Art, Heritage and the Gaeltacht, 2014). The document also addresses legislation and regulations regarding the impact on marine mammals and briefly discusses underwater acoustics as well as estimating and managing risks. But it lacks information about sound emission standards and recommendations on control programmes for underwater noise. Other documents are available containing guidelines and risks for high impulsive sound sources, but they are for airguns used in oil exploration.

For a few years now, the **United States** has had guidelines containing the basic principles for underwater acoustics, measurement methods and mitigation techniques for pile driving noise (Oestman et al., 2009). In addition, the guidelines cover the impact on fish. ANSI does not at present have any standard for measuring pile driving noise, although it does provide a terminology standard (ANSI/ASA S1.1-2013).

The technical guidelines and guidance in the documents described above primarily cover the following topics:

- Metrics and units
- Measurement methodology
- Equipment specifications
- Calibration
- Data management
- Data analysis
- Documentation

A new area within underwater acoustics deals with taking measurements and presenting data for a distance of 750 m for pile driving noise. For other sound sources, you specify the source level by counting back to 1 m. This "trick" is used to avoid the near-field dilemma as described above for pile driving noise and to provide continuity with historical data. German guidelines indicate that measurements should be made with at least three hydrophones at different distances: at 750 m, in the nearest Natura 2000 area and 5 km from the pile driving activity. When construction work begins, noise level measurements should be made in real time and analyses of data in the near future (within a day or two). If none of the factors that affect the acoustic conditions or construction work are changed, it is sufficient to perform the measurements and perform the analyses afterwards. The precise formulation of the measurement programme is subject to the license issued by the Bundesamt für Seeschifffahrt und Hydrographie (BSH). A similar recommendation for measurements during the initial construction work as well as at various distances is available in the Netherlands, but they require one fixed measurement station and one movable, such as onboard a ship. In Great Britain, the levels of underwater noise radiated from the first four installations of the wind turbine foundations must be monitored. If the measured noise levels exceed the levels from the environmental impact assessment, work may not continue without additional permission from the licensing authority.

Recommendations for reducing radiated noise using mitigation techniques are recommended by virtually all countries, and some documents describe how this should be done. More on this in Chapter 4.

Some of the documents address acoustic modelling and recommend that it be done during the permit process to get a first idea of the radiating noise from pile driving. But they also recommend performing sound propagation measurements in situ to gain better modelling accuracy and thus a better estimate of the potential impact on marine life.

# 4 Sound propagation and models

To gain an understanding of how sound propagates, measurements can be supplemented with acoustic modelling. This section contains an overview of the factors affecting sound propagation in water and seabeds. It also provides a few specific examples of these pile driving factors in which acoustic models are used to estimate the noise level of a hammer stroke at different distances in different environments. Worth considering is that the modelling is done using the sound's reduction as the starting point. Therefore, the source level indicated is of major importance. For an in-depth reading on sound propagation in Swedish waters, we recommend Jensen et al. (2011) and FMV (2013). As mentioned before, some countries recommend modelling to estimate the radiated noise for the specific sites of the operation during the permit process. This way, one can gain a better picture of the scale of a potential disturbance at an initial stage as well as the mitigation techniques that may be appropriate.

## 4.1 Sound propagation in Swedish waters

Swedish waters consist of the North Sea and the Baltic Sea. The North Sea consists of the Skagerrak to the north (average depth 210 m), which is connected with the North Sea and is thus saltier (30–35 PSU) than the Kattegat to the south (18–34 PSU and average depth 23 m). The absorption of sound energy in the water is dependent on the salinity, which allows higher frequencies (> 5 kHz) to be absorbed more strongly in the North Sea than in the Baltic Sea and the Bothnian Sea. The Baltic Sea's average depth is 57 m, and salinity varies between 4 and 13 PSU.

How temperature and salinity vary with depth is crucial to how far sound propagates in the water, since these determine the so-called sound velocity profile (SVP) in the water. Information on sound velocity in Swedish waters is available in databases owned by the Swedish Meteorological and Hydrological Institute (SMHI). If, for example, SVP has a minimum midpoint in the water column (occurs in the summer in the Baltic Sea when the surface water is warm and heavier salt water lies at the bottom), then the sound is focused in a sound channel around this minimum because sound waves are refracted in the direction where the velocity is lower (Figure 19).

The higher the frequency, the better the sound is encapsulated in the channel, and can thus be propagated very far before it attenuates. This happens at longer distances in the Baltic Sea than in the North Sea due to the low salinity of the Baltic Sea. For frequencies higher than 5–10 kHz, the absorption in the water (which increases with salinity) is no longer negligible.

However, at lower frequencies the sound leaks out of the channel and starts to interact with the surface and bottom of the water, with increasing transmission losses. How quickly the sound dies out depends on the characteristics of the bottom substrate. If the substrate consists of muddy sediments, then the sound dies out much faster than if it were to consist of hard rocks.

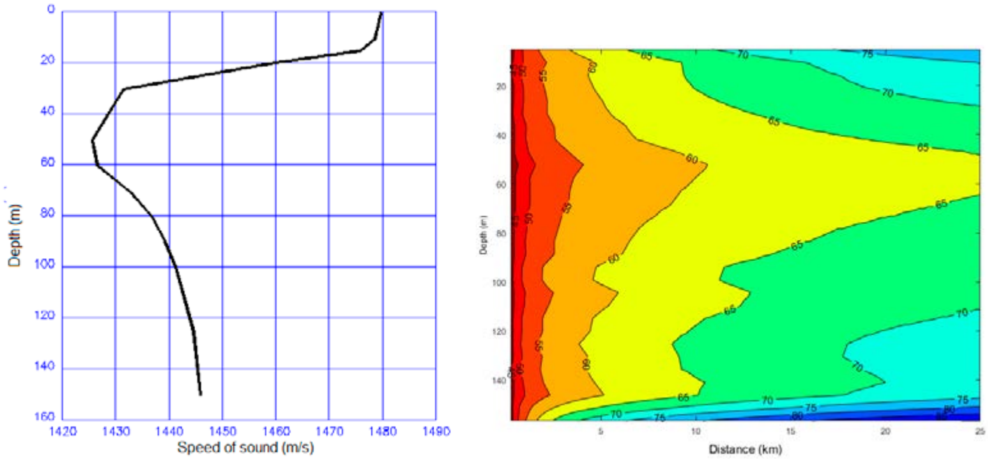



Figure 19. Example of transmission loss in the southern Baltic Sea in August for a 100 Hz frequency. Left: The sound velocity profile with a distinctive sound channel around a 60 m depth. Sound is captured in this channel and propagates much farther than beyond the channel. Right: Transmission loss in dB re 1 m as a function of distance (that is, not received sound level).

Because Swedish waters are relatively shallow, sound strongly interacts with the seabed. The sound velocity profile differs over the year and can vary around Sweden's coastline. Sound velocity in the water is generally higher in the saltier Kattegat than the southern Baltic Sea (Figure 20). This relationship has great signficance for how quickly the pile driving noise will be mitigated.

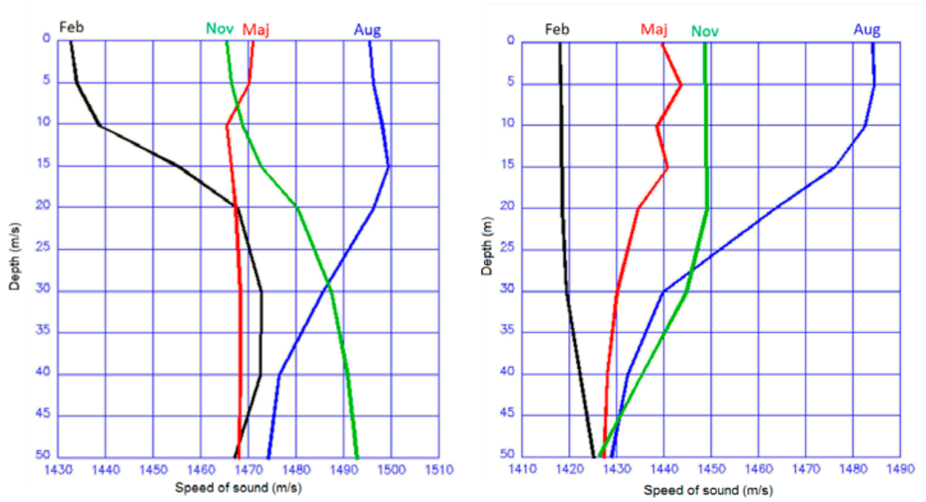



Figure 20. Example of monthly averages (from SMHI) of sound velocity profiles in the Kattegat (left) and in the southern Baltic Sea (right). Note the different scales on the x-axis.

Before computers existed, simplified methods were used to estimate transmission loss (TL). One of these is $TL = klog_{10}(r) + \alpha r$, where $\alpha$ is a measure of the mitigation in the water due to salinity, $r$ is the distance and $k$ is a number between 10 (known as cylindrical propagation) and 20 (known as spherical propagation). At distances up to approximately 5 times the water depth, the propagation is essentially spherical. For greater distances, $k$-values between 15 and 18 are usually used for the Baltic Sea. This estimate of transmission loss is quite rough and applicable only for longer distances (several kilometres), which is why it is not used in practice other than for summary calculations.

## 4.2 Sound propagation models

Today, numerical sound propagation models are used to calculate sound propagation in water. These models require the input of water depth, salinity and SVP (obtained, e.g., through measurements with CTD sensors that register conductivity, temperature and depth vertically in the water column), as well as density and mitigation in bottom sediments and bedrock (harder to measure and estimate, but there are bottom mappings for some areas created by the Swedish Geological Survey, SGU). Existing models can roughly be divided into two classes: ray-tracing based and wave-equation based. In the first class, sound propagation is represented by rays, whose paths are determined by how the sound velocity varies in the media (analogous to geometrical optics). This type of model is applicable for high frequencies, typically higher than 1 kHz. The models are fast, and the ray traces also give a clear picture of how sound propagates in the media. Because high frequencies do not penetrate more than the top part of the seabed, it is generally sufficient to provide input on the surface layer properties for the model.

For lower frequencies (below 1 kHz), ray-tracing models can be insufficient. For example, they do not handle diffraction, in which sound leaks into acoustic shadows where rays do not reach (compare light (high frequency) and sound (low frequency)). We cannot see a person standing behind a corner, but we can hear the person talking. Acoustic shadows can occur when the water depth varies (deep water after shoaling) or due to the prevailing sound velocity profiles as described above. To deal with the wave nature of sound you have to use a wave-equation-based sound propagation model. Because low-frequency sound penetrates the bottom to a greater depth, these models need a more elaborate description of the characteristics of the bottom substrate than the ray-tracing models. Due to the high penetration depth, low-frequency modelling is used for activities like oil exploration when searching for oil reservoirs.

There are several sound propagation models that are open for everyone to use. They are available at the Ocean Acoustics Library (http://oalib.hlsresearch.com/), a website that provides the code and data for modelling sound propagation in water. The library contains codes that have been developed in different parts of the world, such as the ray-tracing code BELLHOP and the wave-equation-based models KRAKEN and RAM. It is important to mention that no single model provides an effective, applied solution for all scenarios. All models have advantages and disadvantages (uncertainties) in relation to their suitability: bandwidth, water depth, computational requirements and ability to account for spatial variability in the environment (Jensen et al., 2011).

In Sweden, the Defence Research Agency (FOI) has developed several computer programmes for underwater sound propagation. Examples include: the ray-tracing codes RAYLAB, MULTIMOC and REV3D; the wave-equation-based RPRESS codes, a full-field model for environments with constant water depths and sediment thicknesses; MODELOSS, a simplified version of RPRESS that calculates only the far field; and JEPE, a far-field model for medium-hard bottoms and moderately variable water depths so that retro-reflected sound can be neglected. Of these, RPRESS is available at the above website.

It is important to choose an appropriate model for a given scenario, but a suitable model can only be predictive if the input maintains the right quality and if there is sufficient spatial and temporal resolution. In fact, the quality of the input sets the limits for the usefulness of the sound propagation calculations. Marine environmental data is often costly to collect, and existing data can be limited. To validate the model, field measurements should also be made. These can help to reduce the uncertainty in the estimated parameters, which is important to be aware of in order to understand the usefulness of the results.

There are a number of commercial Internet-based software tools (platforms) that gather various sound propagation models together with certain environmental parameters such as bathymetry, sediment and sound velocity profiles for specific areas. Some platforms can also have biological parameters like propagation maps for certain animal species. These are usually produced by individual companies to make environmental impact calculations for various sound sources. Examples of how these platforms work and their results are given in Shuy and Hillson (2006), Folegot (2010), Kongsberg (2010) and MacGillivray et al. (2011).

## 4.3 Modelling of pile driving noise

The following section contains examples of how numerical sound propagation models can be used to calculate the levels of noise from pile driving operations associated with constructing offshore wind farms. The examples illustrate sound propagation at two different sites representing the different sea areas along the Swedish coast: Kattegat in the North Sea and a site in the southern Baltic Sea. The precise location cannot be specified because this is a simulation of what sound propagation looks like in two study areas. Furthermore, the calculations are made during two different times of the year (February and August) to exemplify both the geographical environment and the seasonal impact on noise levels. The results of this type of calculation can be used to determine at what time of year pile driving work should be carried out to minimise the negative environmental impact of the noise as much as possible.

### 4.3.1 Sources

To describe a realistic pile driving source for the propagation calculations, data from an unmitigated pile driving operation in the German North Sea is used (data recorded by the consultancy ITAP in Germany). The depth at the measurement site was 20 m, and a steel pile with a diameter of 6 m was driven into a sandy bottom with a hydraulic hammer with an impact energy of 700 kJ. The measurement was made 750 m from the driving, and at the time of measurement there were no mitigating measures in place. In this case, most of the energy was found at around 80 Hz. The calculated sound exposure level, summed over the entire frequency spectrum 10 Hz to 25 kHz, was 177 dB re 1 µPa²s. This level is in line with other measurements made at similar pile driving operations in the area and with the same pile size (Figure 16).

Because the sound propagation models are based on a source level of one metre as input, the transmission loss is first calculated using the environmental parameters prevailing at the time of measurement in the North Sea. It was assumed that the bottom consisted of a 40 m thick sediment of clay-mixed sand on top of solid bedrock, and salinity was assumed to be 34 PSU. Furthermore, a SVP from the time of measurement was used. For frequencies < 800 Hz, the wave-equation based sound propagation model JEPE was used, and for frequencies ≥ 800 Hz the ray-tracing model MULTIMOC was used. A calculation was made for each third-octave band in SEL dB re 1 µPa²s, whereupon the source spectrum at one metre was obtained by adding the estimated transmission losses to the measured third-octave band spectrum at 750 m (Figure 21). This gave an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s at one metre. Notable is that this is not the actual source level at one metre but the equivalent level at this distance.

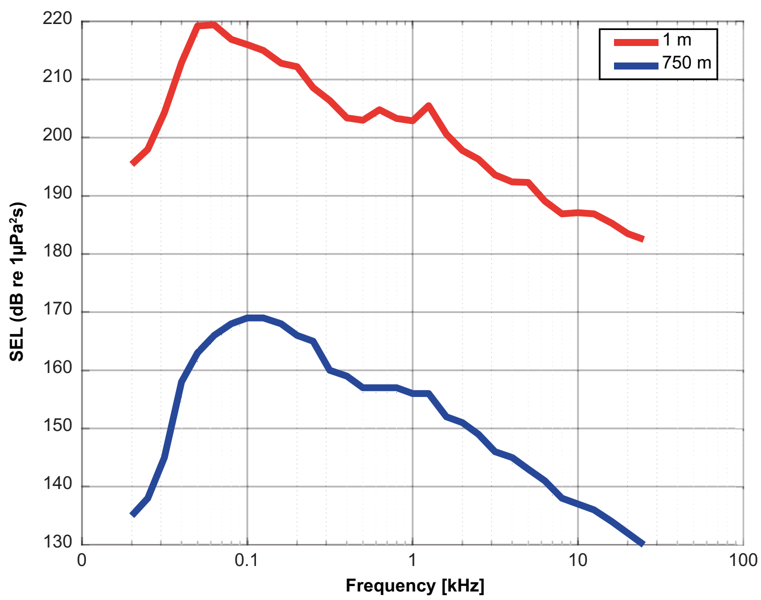



Figure 21. Third-octave band spectrum ($SEL_{(ss)}$) for a pile strike measured in the German North Sea at 750 m from the pile driving source (blue line), and the corresponding levels calculated for 1 m distance in a typical North Sea environment. The depth at the measurement site was 20 m and the steel pile had a diameter of 6 m. Impact energy was 700 kJ and the pile was driven into the sandy bottom without any mitigating measures. Data from ITAP in Germany.

No good models currently exist for calculating the actual source level in the near field, as this is quite complicated for this type of sound source. This source spectrum was then used to simulate the sound propagation in the two selected Swedish areas using the above-mentioned sound propagation models (at the same frequencies).

### 4.3.2 Environmental models in the two sample areas

When calculating sound propagation in the southern Baltic Sea (near the Midsjö banks), the bottom was assumed to consist of a 20 m thick sediment of sand on top of bedrock. The same bottom was also used for the Kattegat calculations (at the height of Falkenberg), with the only difference being that a 0.5 m thick sediment of watery mud was laid on top of the sand sediment which is common here.

The salinity in the southern Baltic Sea was set to 7.5 PSU and at Kattegat to 34 PSU. The water depths were obtained from the Swedish Maritime Administration's database and ranged between 20 and 40 m. To access the highest resolution data available for the seabed composition in Sweden, one can contact the Swedish Geological Survey. The calculations were made in 32 radial sectors out to a distance of 25 km. First, the transmission loss of the noise level was presented as a function of distance in a given bearing and for the various third-octave bands. Furthermore, to make the results more easily understandable, they are presented only for two third-octave bands when the results are plotted for a geographical area. When results for the 100 Hz third-octave band are presented, they refer to the energy contained in the third-octave band 89.1–112 Hz with the centre frequency 100 Hz, and the third-octave band 1,778–2,239 Hz for the centre frequency 2 kHz. The equivalent source levels used for 100 Hz and 2 kHz were, respectively, $SEL_{(ss)}$ 216 dB re 1 µ$Pa^2$s and 197 dB re 1 µ$Pa^2$s . To also show differences in the propagation's effect on the cumulative SEL level for a pile strike for the entire spectrum (10 Hz to 25 kHz), these calculations were also made with an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µ$Pa^2$s.

### 4.3.3 Results of the sound propagation simulations

FREQUENCY-DEPENDENT PROPAGATION

Calculated third-octave spectra for the Kattegat (Figure 22) and southern Baltic Sea (Figure 23) are shown at four different distances from the source. As the figures show, the transmission losses are greater in the Kattegat (the noise level decreases) for the higher frequencies (> 1 kHz) due to the higher salinity. At the long distances calculated (up to 25 km) in these examples, the bottom depths vary greatly, having a major impact on the estimated noise level. If a different bearing had been chosen, the results would probably look different. At the other end of the spectrum bands, we can note that the noise level becomes lower altogether for frequencies < 40 Hz (southern Baltic Sea) and < 30 Hz (Kattegat). This is because sound cannot propagate in a channel (water column and sediment) below a certain limit frequency, the so-called cut-off frequency. The sound velocity profile also affects the radiated noise (see next paragraph for a clearer demonstration).

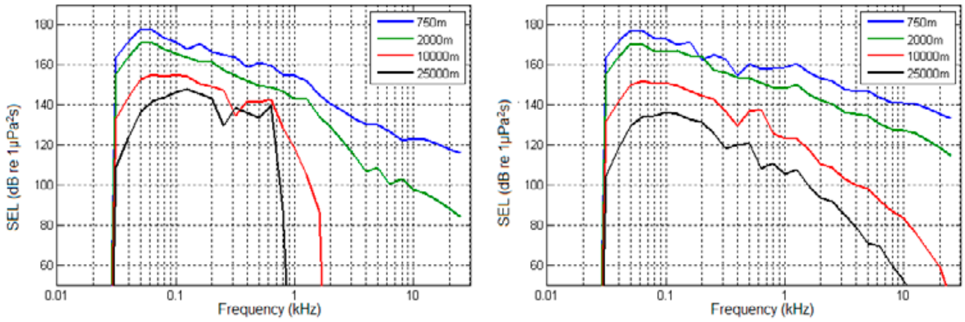


Figure 22. Sound exposure level ($SEL_{(ss)}$) in third-octave band levels for a pile strike calculated for the Kattegat at different distances from the source along the bearing 45–225° in February (left) and in August (right). Sounds with low frequencies are mitigated completely because of the insignificant depth. For sound velocity profiles for February and August, see Figure 20.

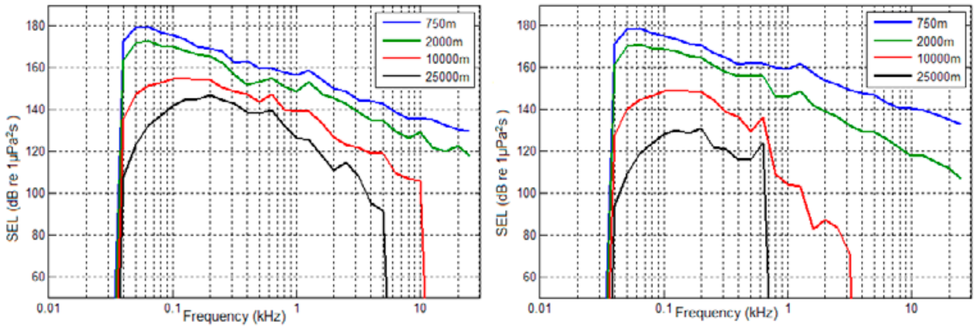


Figure 23. Sound exposure level ($SEL_{(ss)}$) in third-octave band levels for a pile strike calculated for the southern Baltic Sea at different distances along the bearing 90–270° in February (left) and in August (right). Sounds with low frequencies are mitigated completely because of the insignificant depth. For sound velocity profiles for February and August, see Figure 20.

## EFFECT OF THE SOUND VELOCITY PROFILE

Figure 24 shows a typical sound velocity profile (SVP) for February in the southern Baltic Sea when the water has no stratification; instead, the sound velocity is largely the same throughout the water column. Figure 25 shows a typical SVP for August in the southern Baltic Sea where the surface water has warmed up and the velocity in this layer is thus higher. The figures also show calculated $SEL_{(ss)}$ as a function of depth and distance for the third-octave band around 100 Hz at the same location and the two months. What is apparent is that the noise levels are lower in August because the SVP refracts the sound down toward the bottom, where it is absorbed to some extent. In the right-hand images in Figures 24 and 25, the seabed is marked with a dashed line in the lower part. In the case of a relatively soft bottom, as described above, the sound will continue down into the seabed. If there is harder material further down, the sound is reflected back up into the water column again.

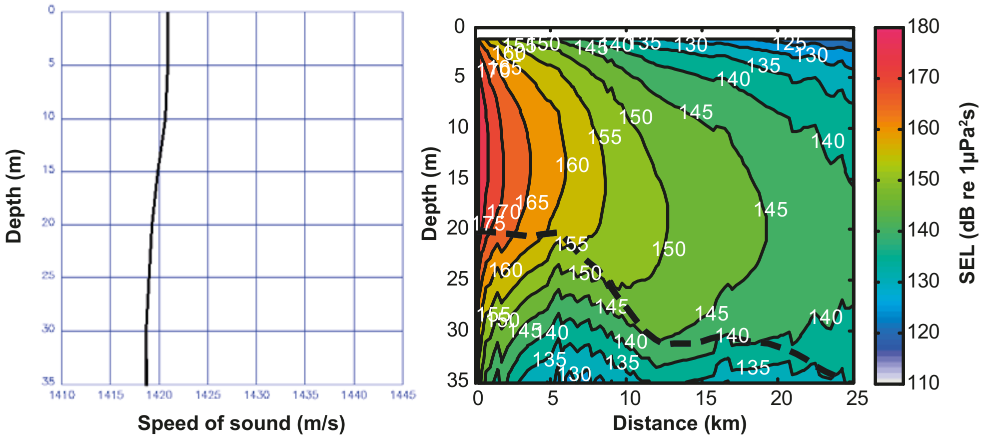



Figure 24. Left: Typical sound velocity profile for the southern Baltic Sea in February. Right: Sound exposure level ($SEL_{(ss)}$) for a pile strike as a function of depth and distance for the third-octave band 100 Hz in August at the southern Baltic Sea in February with an equivalent source level of $SEL_{(ss)}$ 216 dB re 1 µPa²s. The seabed is marked with a dashed line.

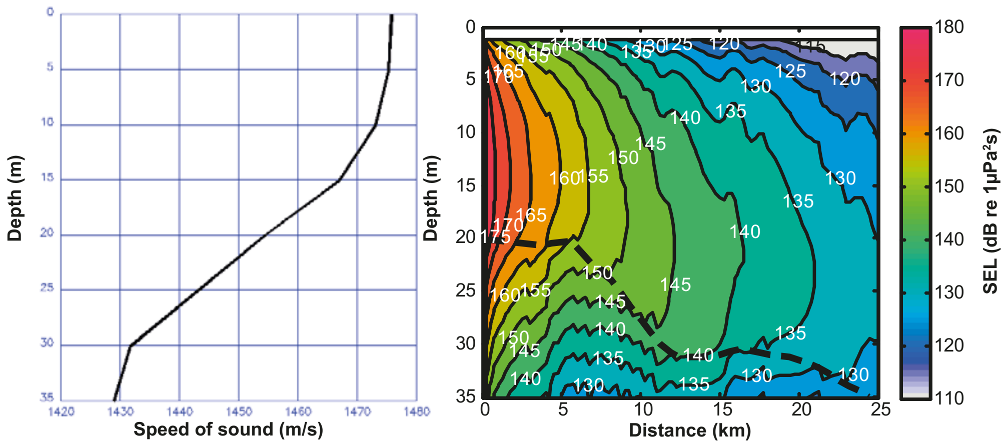
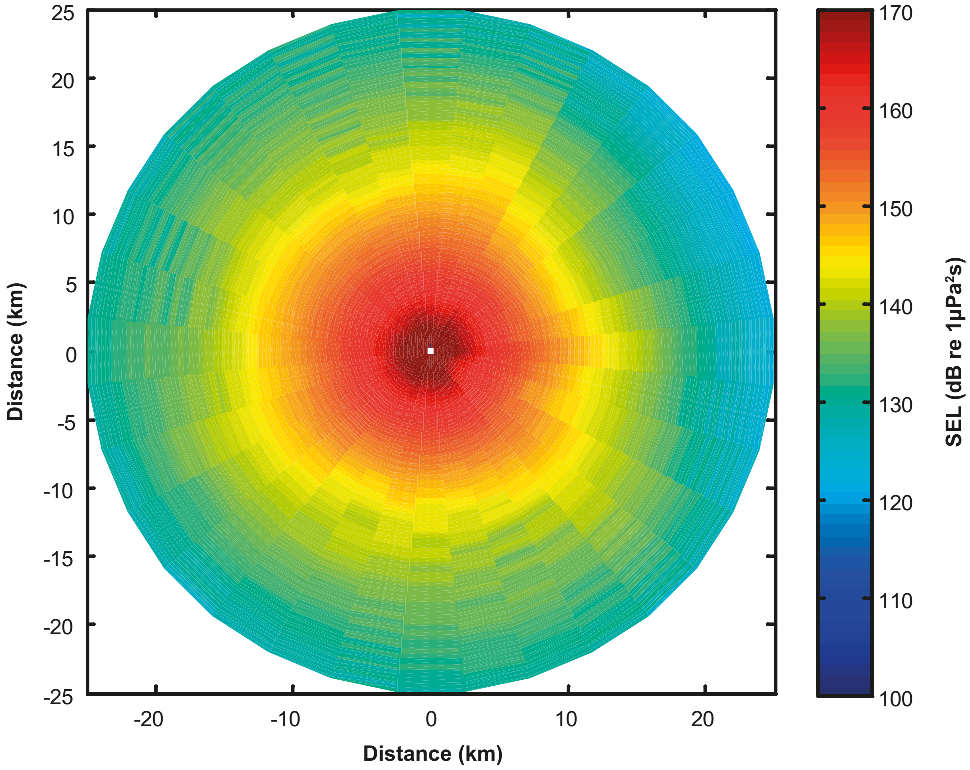



Figure 25. Left: Typical sound velocity profile for the southern Baltic Sea in August. Right: Sound exposure level ($SEL_{(ss)}$) for a pile strike as a function of depth and distance for the third-octave band 100 Hz in August. As apparent, the noise levels are lower in August than in February with an equivalent source level of $SEL_{(ss)}$ 216 dB re 1 µPa²s. The seabed is marked with a dashed line.

## SPATIAL DISTRIBUTION OF PILE DRIVING NOISE

In Figure 26, calculated sound exposure level is shown as a function of direction and distance for the third-octave band with centre frequency 100 Hz at the southern Baltic Sea and the Kattegat in August. In these examples, the data are plotted on a rose diagram where the sound propagation is calculated for each sector (bearing). The dark blue area in the bearing range [–20° to 135°] (Kattegat) represents land. Noise levels are generally higher in the Kattegat than in the southern Baltic Sea, which is in line with the 10-km and 25-km spectra at 100 Hz in Figures 22 and 23 and depends on factors such as different bottom types and SVP. There is also a relatively large difference in the various bearings (sectors) within each area from the centre point outwards depending mainly on differing bathymetry.



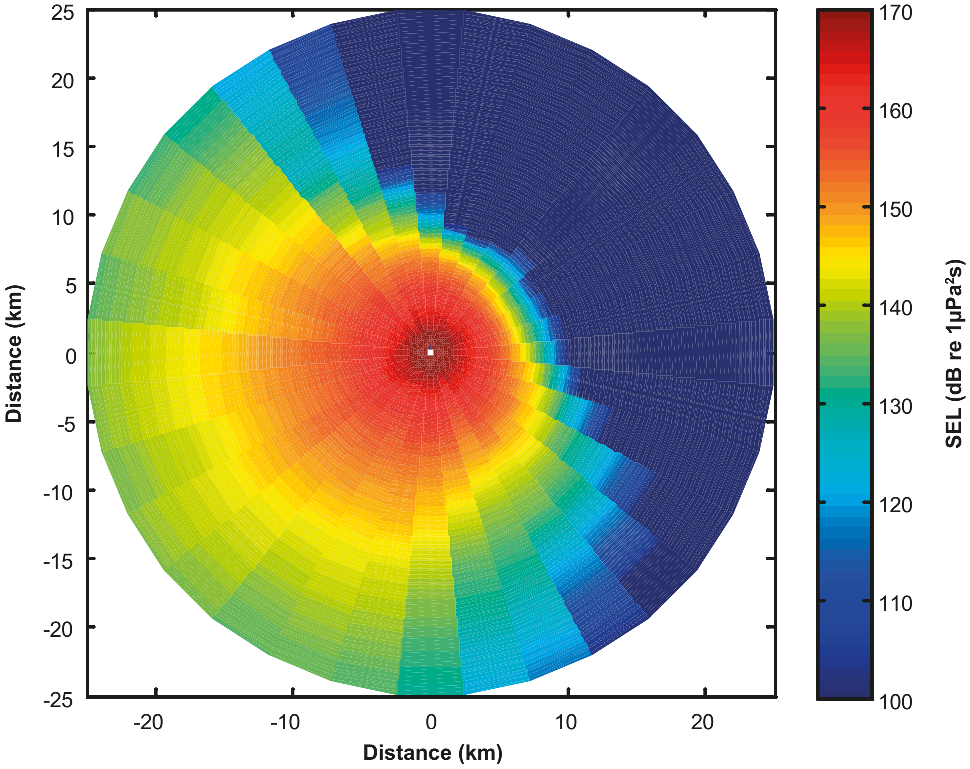




Figure 26. Sound exposure level for a pile strike for the third-octave band around 100 Hz with an equivalent source level of $SEL_{(ss)}$ 216 dB re 1 μ$Pa^2$s as a function of direction and distance in August for the southern Baltic Sea (top) and the Kattegat (bottom), where the dark blue area in the bearing range [–20°, 135°] (Kattegat) represents land.

In Figure 27, calculated SEL$_{(ss)}$ is shown as a function of direction and distance for the third-octave band with centre frequency 2 kHz at the southern Baltic Sea and the Kattegat according to the same principle as for the 100 Hz third-octave band. The noise levels are significantly lower in the Kattegat due to its higher salinity, and are even lower for higher frequencies (compare Figure 22 and Figure 23).

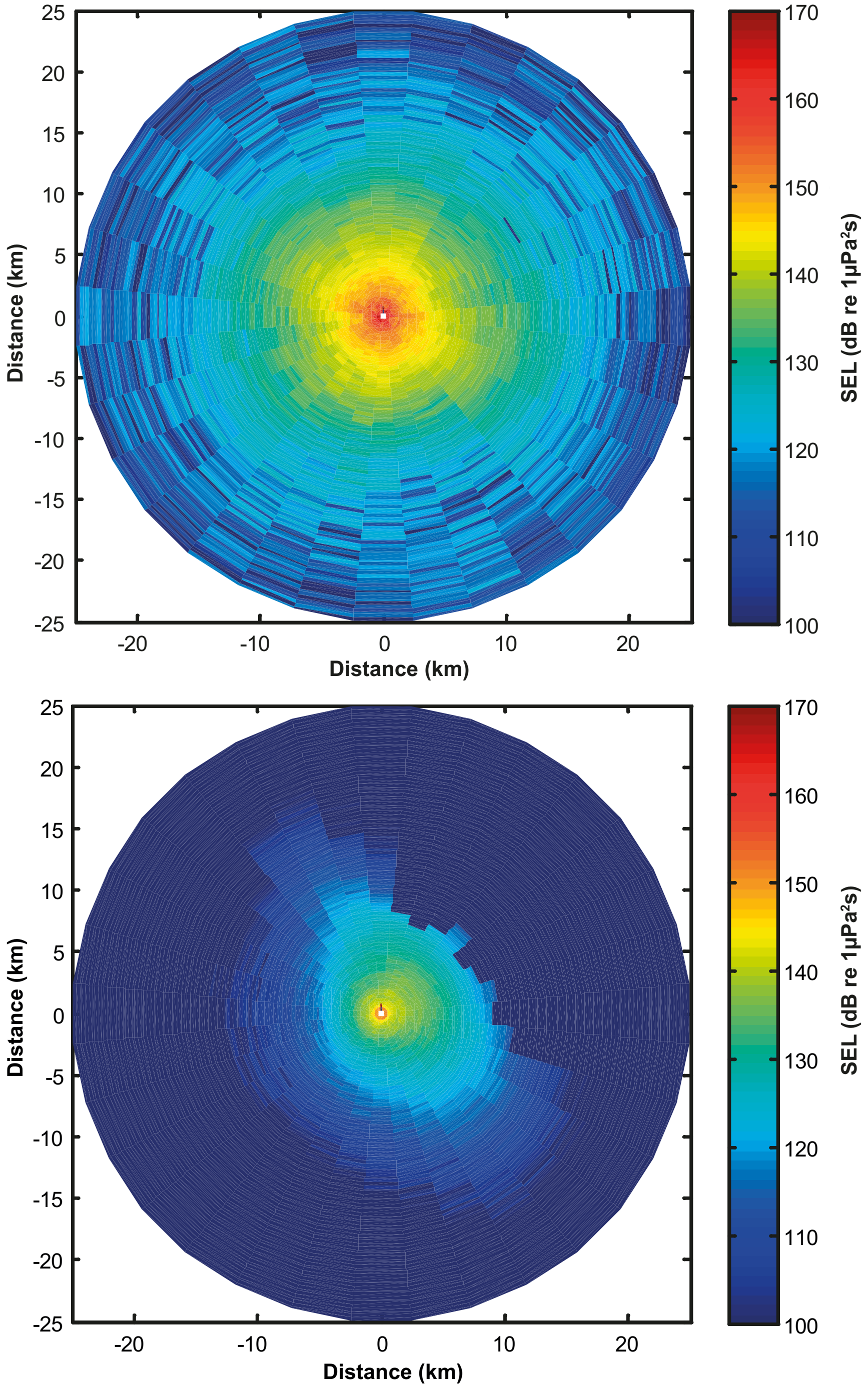



Figure 27. Sound exposure level for the third-octave band 2 kHz with an equivalent source level of SEL$_{(ss)}$ 197 dB re 1 µPa$^2$s as a function of direction and distance in August for the southern Baltic Sea (top) and the Kattegat (bottom) in which the dark blue area in the bearing range [–20°, 135°] (Kattegat) consists of land.

To clarify how the integrated sound energy of the source pulse changes as a function of distance, all third-octave bands (10 Hz–25 kHz) are summed up for the Kattegat (Figure 28). This gives an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa$^2$s. Out to about 3 km, the noise levels for February and August follow each other and are then reduced further in August as a result of a change in sound velocity profile. The irregularity of the noise level in August is due to a downward-refracting sound velocity profile, which leads to a strong interaction with the bottom at the same time as the water depth varies. Results should be seen as an illustration of how sound propagation can vary in a similar area. Making a more accurate estimate prior to construction requires accurate information about bottom parameters in particular, and the use of the probable SVP for the intended construction period.

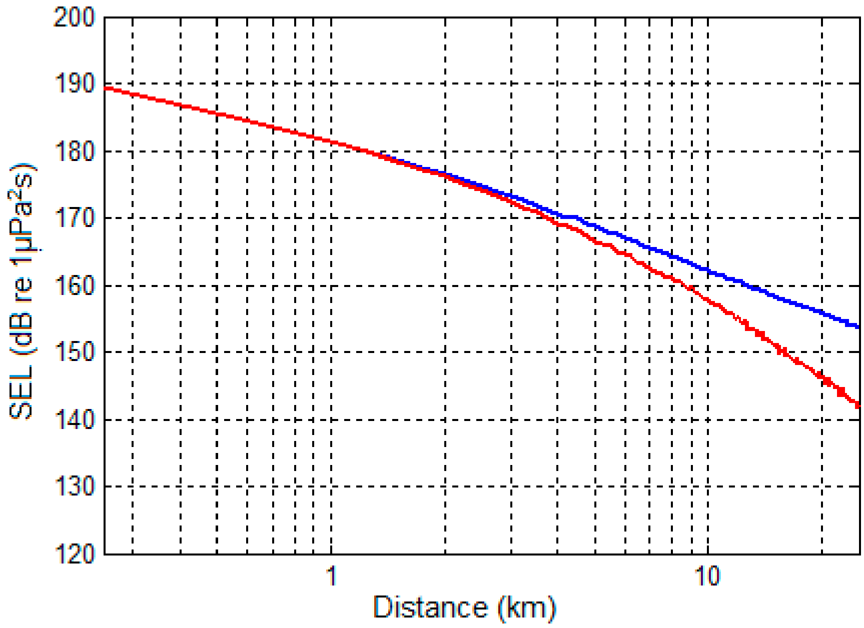



Figure 28. Sound exposure level summed over all frequencies (10 Hz–25 kHz) in the source spectrum, which gives an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa$^2$s as a function of distance along the bearing 45–225° in the Kattegat. The calculations are made with sound velocity profiles typical for February (blue) and August (red).

# 5 Noise mitigation methods

Research on noise mitigation associated with pile driving has been ongoing for several years (CSA Ocean Sciences Inc., 2014; OSPAR, 2014), in part because of Germany's threshold values for harbour porpoise injuries. The choice of technique, impact energy and pile diameter give an indication of the noise levels that can be generated so that suitable mitigation techniques can be designed. Other necessary steps include calculating and estimating the area's specific sound propagation conditions in relation to the period during which the driving will take place. Furthermore, it is important to take an inventory of the presence of marine animals and activity in conjunction with the planned pile driving operation to determine mitigation requirements. These factors should be compiled in order to plan for the area's specific mitigation requirements with regards to protecting wildlife in the pile driving area.

As a rule of thumb, one can assume that a 20 dB reduction in pile driving noise represents a 90% reduction of sound pressure and 99% reduction of sound pressure intensity. One should remember that all the different kinds of mitigation techniques will result in a variety of possible reductions achieved. Moreover, the selected technique depends on which frequencies are primarily mitigated. There is also very little experience of water depths greater than 40 m. At such depths, it becomes much more difficult to achieve adequate mitigation.

The following sections contain an overview of the most common noise mitigation systems in use today. Additional techniques exist but are mainly at the experimental stage.

## 5.1 Bubble curtains

Bubble curtains have long been deployed to mitigate radiated noise and represent a well-proven technique (Würsig et al., 2000; Lucke et al., 2011; OSPAR, 2014). A bubble curtain is formed by releasing bubbles at the bottom using compressed air that is pushed through a perforated hose. Because air and water have a substantially different acoustic impedance, the noise is mitigated when the sound is forced to propagate through the bubble curtain. By creating different bubble sizes and increasing the flow of air, the noise-reducing effect increases (Elmer et al., 2007b; Bellmann, 2014) (Figure 29). It is clear that the bubble curtains used begin to have a mitigating effect at 80 Hz, after which the reduction increases the higher the frequency. This example is typical of what you can expect in terms of frequency-dependent mitigation (OSPAR, 2014).

Bubble curtains are categorised by size and placement related to the noise source. There are big bubble curtains (BBC) and small bubble curtains (SBC). The big BBCs can also have double bubble curtains (DBBC). The SBCs mitigate only the noise from the pile, while the BBCs envelop the entire construction site including the vessel, providing a dampening of radiated noise from the entire operation (Figure 30). Since bubble curtains have been used relatively often, there is much data on their effectiveness. As previously described, many studies have been conducted in Germany. According to a compilation of measured noise levels from different mitigation techniques, the different bubble curtains dampen the radiated noise by 5–18 dB, depending on the number of curtains and sizes (Table 8). The levels are different due to the different conditions, such as currents and water depths, since these factors influence how effectively a bubble curtain stays in place. At high currents, the curtain risks collapsing.

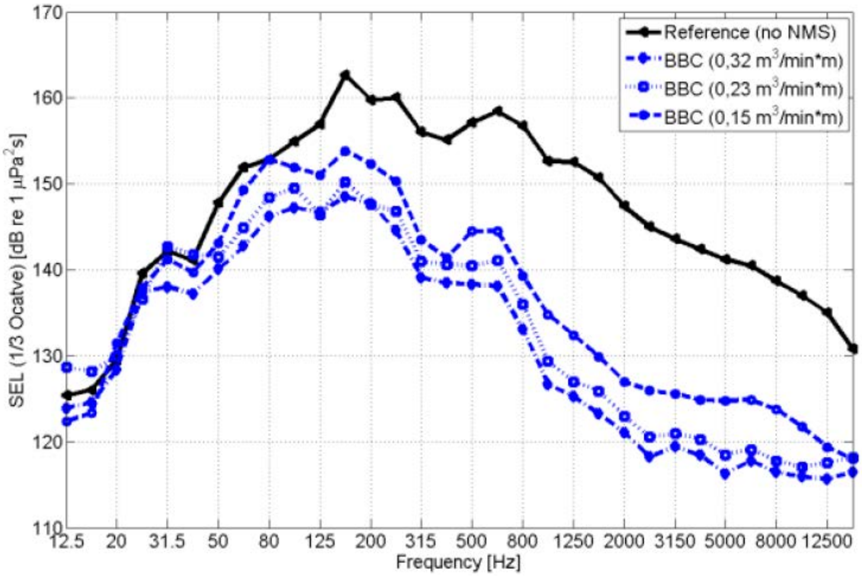



Figure 29. Amount of bubbles/air pressure as a function of sound level (SEL) compared to without a BBC when driving a steel pile with a 2.4 m diameter and with an impact energy of 800 kJ, from Bellmann (2014).

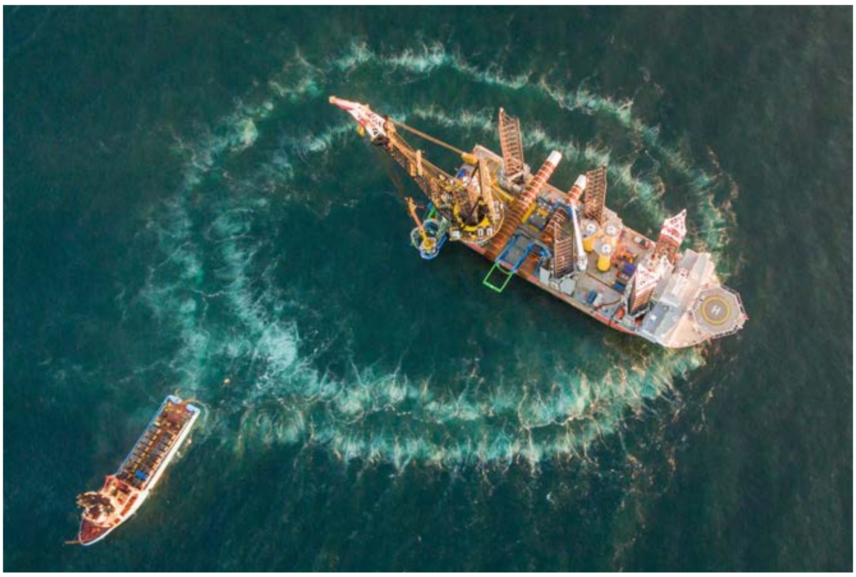


Figure 30. An example of a double big bubble curtain from the installation of a wind turbine foundation (Photo: Vattenfall).

## 5.2 Isolation casings

Isolation casings act by creating a shielding effect, similar to that of noise barriers in the air (see Figure 26). Only a simple steel tube can reduce noise to some extent. The isolation casing is placed at the bottom of the seabed and can be supplemented with pneumatic foam plastic sections and internal bubble curtains. As with bubble curtains this technique exploits the difference in impedance between air and water, so energy is absorbed, distributed and dissipated (Elmer 2007a; Nehls et al., 2007). Isolation casings are reusable and therefore cost-effective as a mitigation system, but they are attached directly to the pile driving system and so affect the time it takes to complete the entire pile driving operation. This results in longer and more expensive installations. Today, there are two main types of isolation casings deployed in full scale: the IHC Noise Mitigation System (NMS) and BEKA Shell. Both work in similar ways, so only IHC-NMS will be described in more detail.

IHC-NMS was developed in the Netherlands and has already been tested in a number of commercial offshore wind farm projects. The system consists of an acoustically decoupled double-wall isolation casing of steel with an air-filled interspace. An adjustable double bubble curtain layered between the casing and the pile provides an additional noise barrier. This system combines the features of an isolation casing with a confined bubble curtain. Many full-scale tests have been performed at different depths and pile diameters as well as for different foundation types (monopile, jacket and tripod). Measurements of noise levels with and without IHC-NMS show that the technique dampens the radiated noise by up to 15 dB (Table 8). However, laboratory studies show that the technique can dampen up to 20 dB (Koschinski and Ludemann, 2013). The technique is considered highly reliable unlike solutions such as bubble curtains, which are greatly affected by local conditions. IHC-NMS dampens the noise already down at 30–40 Hz, but is most effective from about 100 Hz upwards (Bellmann et al., 2015). The internal bubble curtain has proved to be especially effective for the frequency band 500–5000 Hz (Wilke et al., 2012).

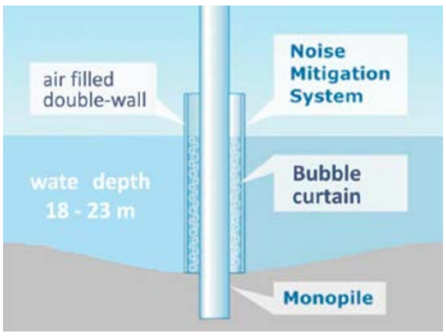



Figure 31. Left: An illustration of the IHC-NMS system with an air-filled double wall that encircles the pile and an internal bubble curtain, from Koschinski and Ludemann (2013). Right: IHC-NMS system in use (the red-white casing) during pile driving for a wind farm in the German North Sea (Photo: Markus Linné, FOI).

## 5.3 Cofferdams

One type of dampening tube is a cofferdam, a massive steel tube placed on the bottom of the seabed. The pile is inserted into the tube and a seal ring at the bottom is installed (Thomsen, 2012). The water is pumped out and the pile is then driven in air. Because of the impedance mismatch between air, steel and water, the radiated noise is mitigated effectively, to a maximum of 20 dB. However, this method has experienced major technical difficulties with keeping the system tight enough (Thomsen, 2012). But even with somewhat leaky seals, noise can be reduced by 10 dB.

## 5.4 Hydro Sound Dampers and encapsulated bubbles

Hydro Sound Dampers (HSDs) use nets with small, air-filled rubber or plastic balloons that are placed around the pile to mitigate radiated noise (Elmer and Savory, 2014) (Figure 32). A similar system called encapsulated bubbles has been developed in the United States (Lee et al., 2012). The advantage of these systems compared to released bubbles is that the size of the bubbles can be designed to mitigate specific frequencies (Lee et al., 2011, 2012; Elmer and Savory, 2014). The greatest mitigation has been demonstrated for the frequencies 100–600 Hz. These systems are easier to use because they do not require the same logistics as bubble curtains. Lab studies have shown a noise reduction of up to 25 dB (Elmer and Savory, 2014), but in full-scale tests in Germany and Great Britain a noise reduction of 13 dB at best has been achieved, with an average of 10 dB (Table 8).

Figure 32. Hydro Sound Dampers (HSD) net with rubber or plastic bubbles (Photo: Vattenfall).

All in all, a large variety of noise mitigation systems are available, some of which we have presented here. They all provide a reduction of 5–20 dB (Table 8). A few of the techniques, such as bubble curtains and IHC-NMS, have been used at more than 100 pile installations (mostly in Germany and the Netherlands) and have proven to be reliable techniques despite some limitations. A combination of at least two systems provides the greatest mitigation. When pile driving using a combination of bubble curtains and IHC-NMS, the noise has been reduced by a maximum of 23 dB.

**Table 8. Overview of noise mitigation techniques and measured mitigation (in dB), and the number of piles that these techniques have been used for. Table modified after Bellmann et al. (2015).**

| Noise mitigation technique | ΔSEL [dB] | Number of test (piles) |
|---|---|---|
| Big bubble curtain (BBC)<br>(>0.3 $m^3$/(min*m), ballast chain inside, water depth <30 m) | 10 < 13 < 15 | >150 (>300) |
| Double big bubble curtain (DBBC)<br>(>0.3 $m^3$/(min*m), ballast chain inside, water depth <30 m, distance between hoses > water depth) | 14 < 17 < 18 | >150 (>300) |
| Small bubble curtain (SBC)<br>(Use air volume, hole configuration) | (5 <) 10 < 14 | 2 |
| Hydro Sound Dampers (HSD)<br>(Number and size of HSD elements) | 8 < 10 < 13 | >50 |
| Noise mitigation screen (IHC-NMS) | 10 < 13 < 15 | >140 |
| Cofferdam<br>(Function of sealing gasket) | problem < 10<br>no problem ≥ 20 | >10 (>10) |
| Combination of two BBC systems (DBBC + BBC) | 15 < 16 < 19 | >30 (>70) |
| Combination of IHC-NMS + BBC | 17 < 19 < 23 | >90 |
| BBC (HTL) + HSD | 15 < 16 < 20 | >10 |
| DBBC (Weyres) + HSD | 14 < 16 < 22 | 2 |

## 5.5 Impact of noise mitigation systems on noise level at longer distances

Finally, we study how the total energy of the source pulse (i.e. energy totalled over all frequencies in the source spectrum) propagates, and what the effect will be of shielding the source with some kind of noise mitigation system. SEL for the propagating pulse as a function of direction and distance with and without a noise mitigation system during August is shown in Figure 33 (southern Baltic Sea) and Figure 34 (Kattegat). In these cases, we assume that the mitigation system reduces the source by 20 dB. This is a probable level of mitigation that has been demonstrated in Germany. As described earlier, it is worth bearing in mind that modelling should be done using the sound's transmission loss as a starting point. Thus it is extremely important which source level is indicated. In this case, we use an equivalent source level of $SEL_{(ss)}$ 226 re 1 $\mu Pa^2s$ at 1 m. For the mitigated scenario, an equivalent source level was used of $SEL_{(ss)}$ 206 re 1 $\mu Pa^2s$ at 1 m. This means that even the received noise level at a given distance and direction will be 20 dB lower than without mitigation.

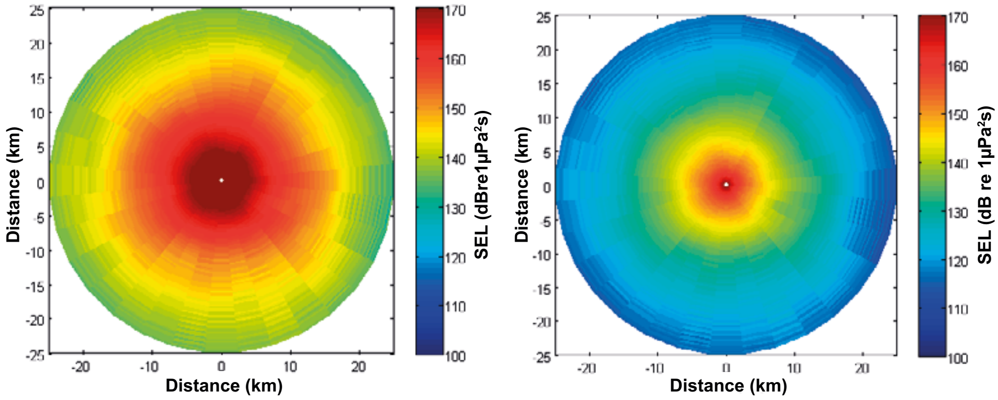



Figure 33. Sound exposure level for frequencies 10 Hz–25 kHz, which gives an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s as a function of direction and distance in August at the southern Baltic Sea. Left: Levels at pile driving without mitigation. Right: Levels with a reduction of 20 dB.

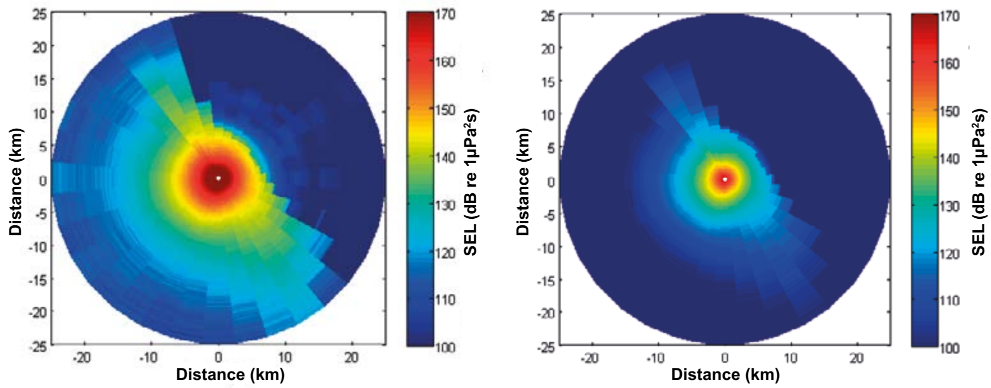



Figure 34. Sound exposure level for frequencies 10 Hz–25 kHz, which gives an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 µPa²s as a function of direction and distance in August in the Kattegat. Left: Levels at pile driving without mitigation. Right: Levels with a reduction of 20 dB. The dark blue area in the bearing range [–20° to 135°] in the images represents land.

To clearly understand how the integrated energy of the source pulse summed over all frequencies changes as a function of distance, data are presented even in a specific direction. In Figure 35, data from Figures 33 and 34 along the bearing 225° from the centre have been extracted to give a picture of how $SEL_{(ss)}$ for the frequencies 10 Hz–25 kHz vary for different distances. We note that the noise level is higher in the southern Baltic Sea generally for all distances, which is probably due to the difference in seabed composition and bathymetry but also sound velocity profile. The effect of noise mitigation measures is clear. If the source is mitigated by 20 dB, the distance for the proposed hazardous level for harbour porpoises, for example, of $SEL_{(ss)}$ 164 dB re 1 µPa²s decreases from approximately 6 km to 600 m in the southern Baltic, and from 1.5 km to 350 m in the Kattegat. This means that the potential adverse affects on the area can be reduced quite substantially if a mitigation system is put to use. Note that the calculated results are based on type bottoms that can be present in these areas. A more detailed bottom survey should be done before the actual sound propagation can be assessed.

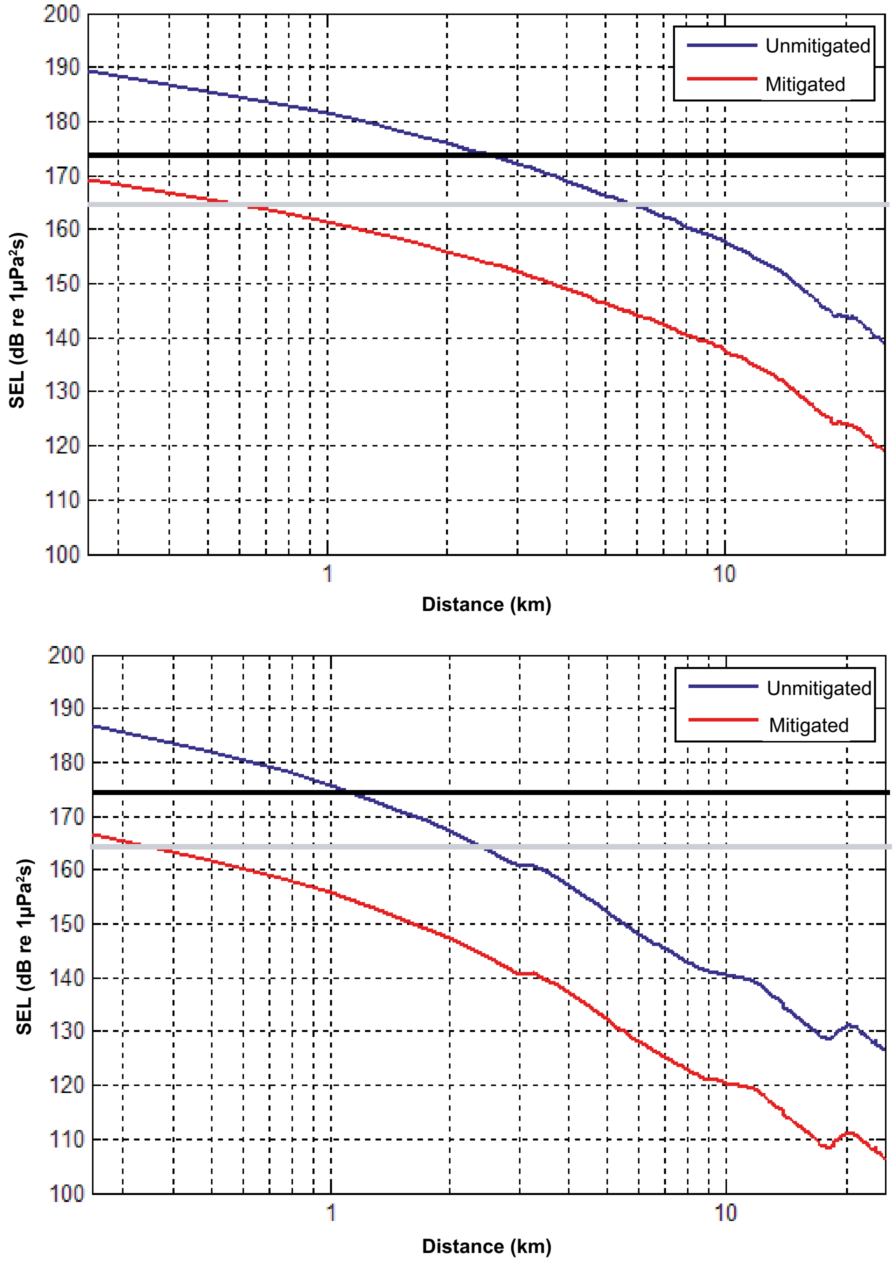



Figure 35. Sound exposure level for frequency 10 Hz–25 kHz, which gives an equivalent source level of $SEL_{(ss)}$ 226 dB re 1 μPa²s as a function of distance along the bearing of 225° in August at the southern Baltic Sea (top) and in the Kattegat (bottom) for unmitigated (blue) and mitigated (red) pile driving. The grey line shows the recommended threshold for TTS for harbour porpoises ($SEL_{(ss)}$ 164 dB re 1 μPa²s), and the black line mortality and injury to the internal organs of fish ($SEL_{(ss)}$ 174 dB re 1 μPa²s). Note that the injury also depends on the number of sound pulses over time.

# 6 The effects of pile driving noise on harbour porpoises, cod and herring

## 6.1 Introduction

The starting point of this study is the prevention of injury to the population of certain species in Swedish waters. The study focuses on the harbour porpoise (*Phocoena phocoena*), cod (*Gadus morhua*) and herring (*Clupea harengus*), species that play an important role in the marine ecosystem and whose survival is either threatened or significant for commercial fisheries (Figure 36). The study aims to collect available information on the species' responses to and the effects of loud, impulsive sounds (pile driving, airguns, explosions and active sonar). Injury and flight behaviour are the main focal points, as these reactions can affect the species at the population level.

The harbour porpoise is one of the smallest species of toothed whales and the only species that can be found all year long in Swedish waters. It is found in the Baltic Sea and along the entire west coast of Sweden. The harbour porpoise population in the southern Baltic Sea is defined as a separate population due to genetic and morphological differences compared with their populations in the Danish straitsand the North Sea (Wiemann et al., 2010; Sveegaard et al., 2015). The population in the Baltic Sea has fallen sharply and today consists of about 500 animals (Carlström and Carlén, 2015) that face the risk of extinction unless special measures are taken. The main threat at present for the harbour porpoise is bycatch in fishing nets, but environmental toxins (PCBs) and decreased food supply due to overfishing also pose threats. Harbour porpoises are protected by several organisations. Both HELCOM's (Helsinki Commission 2013) and IUCN's (IUCN 2015) red list of threatened species classify the Baltic population as "critically endangered" (CR), while its population in the Danish straits and the Kattegat is classified as "vulnerable" (VU) according to HELCOM. On the Swedish Red List (Swedish Species Information Centre 2015), harbour porpoises are a stock in Swedish waters that fall within the category of "vulnerable" (VU). Furthermore, the harbour porpoise is also strictly protected under the EU Habitats Directive (Council Directive 92/43/EEC), as it must be protected from both within and outside Natura 2000 sites (*Species protection under the Habitats Directive Annex II and Annex IV*). The harbour porpoise is also listed on the OSPAR list of threatened and/or declining species in the Northeast Atlantic (OSPAR 2008).

Cod and herring are some of the most important fish species in Sweden. Both species occur in all Swedish coastal waters. Cod distribution is, however, limited in the Bothnian Sea by the low salinity. Cod stocks have declined sharply since the 1980s, and generally end up in the category of "vulnerable" (VU) in the Swedish Species Information Centre database (2015) and HELCOM'S (2013) red list. High fishing pressure mainly represents the biggest threat to cod, but lack of oxygen in the bottom water and increased nutrient load also contribute to the decline of the fish stock. Cod is divided into separate stocks that are classified differently in the HELCOM red list. Cod in the eastern Baltic is classified as "vulnerable" (VU) and in the western Baltic as "near threatened" (NT). Cod in the Kattegat, whose spawning biomass has declined 90% since the 1970s (ICES 2012), is considered "critically endangered" (CR).

The harbour porpoise is a predator in the marine food chain and has a relatively high nutrient requirement; its diet consists mainly of cod and herring (Börjesson et al., 2003; Sveegard et al., 2012). Cod is considered to be the most important predatory fish in the North Sea and in large parts of the Baltic Sea, while herring constitutes an important food source for many marine predatory fishes, birds and mammals. Cod is primarily a demersal fish, but adult individuals are also pelagic. Herring are pelagic fish but are substrate dependent during spawning, when they gather in shallow waters along the coast and on offshore banks. Herring eggs remain in the areas where they sink to the bottom and form aggregates. Shallow marine areas (<30 m depth) are also attractive areas for establishing offshore wind farms, and the pile driving work can thus coincide with the spawning season for herring.

In light of the above, harbour porpoises, cod and herring are important species to safeguard during pile driving when anchoring a wind turbine foundation.

Figure 36. The literature review focuses on three specific species in Swedish waters: the harbour porpoise (*Phocoena phocoena*, at top), cod (*Gadus morhua*, bottom left) and herring (*Clupea harengus*, bottom right). (Harbour porpoise and herring photo © Sandra Andersson, Marine Monitoring AB; cod photo © Mathias Andersson, FOI).

### 6.1.1 Hearing of harbour porpoises and fish

Hearing is one of the most important senses for mammals and fish, something that is an effect of the relatively poor visibility under water. Compared with sight and smell, hearing gives an organism an image of its surroundings at long distances and can be used to detect prey and predators.

A natural soundscape of the ocean occurs when sound from abiotic factors (e.g., waves and wind) are mixed with the sounds of biotic factors (e.g., sounds from fish, porpoises, shrimp, etc.). The natural soundscape is amplified by human-generated sounds (anthropogenic sounds) like shipping vessel traffic, seismic surveys and underwater construction. These sounds often lie within the same frequency range as the fishes' or mammals' hearing (Figure 1). As a consequence of the development of renewable energy production, offshore construction is increasing, often with the help of pile driving. The pulses of extreme noise associated with pile driving may cause flight behaviours and physiological harm in species such as marine mammals and fish that are present in the environment (Figure 37).

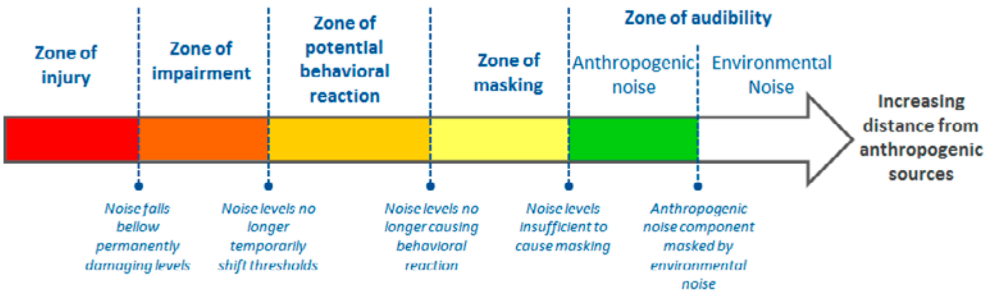



Figure 37. Illustration of the potential effects of noise which demonstrates the relationship between noise, distance and impact; from Dooling and Blumenrath (2013).

Harbour porpoises have a large hearing range that enables them to listen to natural sounds in their environment and to echolocate. During echolocation, animals emit sounds that are then reflected when they hit objects in the water mass. The reflected signal is registered by the porpoises, who generate an "image" of their surroundings. The porpoises use echolocation to find prey and to navigate.

Hearing is well developed in fish, but it differs physiologically from mammals mainly because fish can register particle motion. Sound is mainly detected in the inner ear as a form of mechanical interaction between sensory hair cells and calcium carbonate structures (otoliths). The particle motion in a sound wave causes a relative movement between the otoliths and the hair cells that is perceived as sound. The particle motion can be detected in all studied species of fish (Popper and Fay, 2011), but only fish with swim bladders can register the pressure component of a sound wave. The gas-filled swim bladder reflects the pressure changes in water that are caused when the passing sound wave compresses and expands the water particles. Fluctuations in the swim-bladder volume cause particle motion that can be registered in the inner ear.

The ability of fish to perceive the pressure component of a sound wave differs among species as there is considerable anatomic variation in the proximity of the swim bladder to the otoliths. In herring and carp, there is a mechanical connection between the swim bladder and inner ear, which means that they are more sensitive to sound both in terms of frequency and sound pressure. Salmon, eels and cod have no link between swim bladder and inner ear and therefore have a higher hearing threshold. For the fish to be able to register a sound, the sound level must exceed the hearing threshold by a few decibels, the so-called critical ratio. Ambient noise also greatly determines how well the fish can hear the sound.

### 6.1.2 Injury from pile driving noise at the individual level

Exposure to extremely loud impulsive noise during pile driving may cause fish to die of injuries to the swim bladder and other internal organs (Popper and Hasting, 2009). A fish without a swim bladder is considered to be less sensitive to pressure changes, but there is also variation among fish with swim bladders depending on the type of swim bladder (see Section 6.2.4). A sound wave caused by a pile strike contains rapid pressure changes that affect the different body parts of fish depending on their compressibility. Fish tissue is minimally affected because of its physical resemblance to water. On the other hand, gases – free or dissolved – have a much higher compressibility than water and they compress and expand as the pressure changes (barotrauma). A gas-filled organ like the swim bladder is compressed more than surrounding tissue as pressure increases, and expands more than surrounding tissue as pressure decreases. The compression and expansion of the swim bladder relative to the fish's tissue can lead to tissue damage and even rupture of the swim bladder (Hastings and Popper, 2005). Barotrauma may also cause formation of air bubbles in the blood vessels and organs, which is often deadly.

High noise levels can also cause permanent hearing damage (called permanent threshold shift, or PTS) or temporary hearing loss (called temporary threshold shift, or TTS) in both harbour porpoises and fish. PTS in harbour porpoises involves damage to the sensory cells in the hearing organ, while TTS partly occurs due to the swelling of specific nerve endings of the hearing organ. PTS in fish can include damage to the sensory hair cells or nerve fibres or other tissue damage. In TTS, sensory hair is torn away from the saccular epithelium in the fish's inner ear. The sensory hair can probably be replaced by new hair cells, and the duration and intensity of the sound may affect the regeneration time.

### 6.1.3 Behavioural reaction in response to pile driving noise

High noise levels generated during pile driving can produce two different types of behaviours in harbour porpoises. Panicky flight behaviour can occur if the individual is completely unprepared for the sound. One concern is whether this contributes to separation of a calf from its mother. The most common response of the harbour porpoise is, however, an escape or avoidance behaviour away from the sound source (Skjellerup et al., 2015).

Expected behavioural reactions in fish exposed to high noise levels include avoidance, flight behaviour, fright response, and altered swimming behaviour (Thomsen et al., 2006; Mueller-Blenke et al., 2010). The behavioural response of different species can vary because the hearing threshold varies and because the species can exhibit different flight behaviours. Considerable variation can also be found within species and among individuals depending on gender, age, fitness and motivation. If an area is important enough to the fish's survival or reproduction, the fish can be more tolerant to the sounds (Bejder et al., 2009). For example, for bottom-feeding fish such as cod, the bottom environment can make a good foraging area, spawning site or act as protection against predators. In the pelagic herring, foraging, spawning, overwintering and migration are mostly relevant. For example, different herring respond differently to impulsive noises, from a strong response during overwintering to low during feeding migration (Pena et al., 2013; Doksaeter et al., 2012).

### 6.1.4 Impact on population levels from pile driving noise

There are no direct field studies that address how the negative effects of pile driving noise affect a species at the population level (Popper et al., 2014; Skjellerup et al., 2015). One hypothesis is that it is primarily a negative impact that causes impaired reproductive success that can result in negative effects at the population level.

The extent of the impact depends on factors like population size, life stage, area and degree of impact. Different stocks or populations can be affected locally, and a negative impact on a single individual can have a significant impact if the survival of the stock is seriously threatened (Skjellerup et al., 2015). The reduced stock of harbour porpoises in the Baltic Sea, for example, is especially vulnerable compared to the populations of harbour porpoises in the Danish straits and the Kattegat. The same reasoning can apply to cod stocks in the Kattegat compared with other stocks in the North Sea, for example. A risk analysis in the context of a wind energy project in the Kattegat revealed that the construction phase can entail a significant risk to the endangered cod population. The effect on the population level occurs, however, only if the pile driving work is carried out in conjuction with cod spawning in the area (Hammar et al., 2014). The effects of offshore wind power on the harbour porpoise population in the Kattegat has been simulated using a model, and the results indicate that the harbour porpoise population is not affected by existing wind farms or the construction of two planned wind farms (Nabe-Nielsen et al., 2011).

Hearing impairments in harbour porpoises like TTS and PTS that are caused by high noise levels might lead to an impaired ability to echolocate, restricting their chances to find prey, communicate and navigate. All this is expected to affect individual reproduction as well as survival. Because harbour porpoises live in cold temperate waters, they regularly need to seek food and replenish their energy reserves in order to avoid freezing to death. If the harbour porpoises are scared away from a productive area with no other alternatives, they risk a reduction in fitness. For adult females who are often both pregnant and lactating at the same time, access to productive areas are of major importance compared with other individuals (Carlström and Carlén, 2015).

Damage to fish from high noise levels affects the individual's survival and/or reproduction directly or indirectly, through reduced fitness. A hearing impairment might prevent the fish from communicating, detecting predators and perceiving its environment. A response to stress, caused by a noisy environment, can generate a higher susceptibility to disease and can reduce spawning success (Thomsen et al., 2006; Sierra-Flores et al., 2015). Even an increased mortality of eggs and larvae can affect recruitment for the total stock. A change in behavioural or migration patterns might affect the population if the fish swim away or avoid a preferential area for foraging, spawning or growth, which can potentially adversely affect the fish's reproduction or fitness or the survival of hatchlings.

## 6.2 Studies on fish (cod and herring)

There are currently few studies on cod and herring that have been exposed to impulsive noise. Drawing parallels with studies on other fish species is difficult because the hearing threshold can vary by around 40 dB between species (Chapman and Hawkins, 1973). Studies that address how cod and herring are affected by continuous noise, such as the operational noise from wind farms and vessels, have not been included in this compilation because the properties of a continuous noise are different from those of an impulsive noise.

A few studies are available on the eggs and larvae of cod and herring. However, the difference between the species is assessed as small, especially for eggs, since it is not until the swim bladder is developed in the larval stage that differences are expected to arise. For this reason, a separate section (6.3) is dedicated to eggs and larvae that refers to studies on several different species of fish.

The following section discusses the hearing of cod and herring (6.2.1). Subsequent sections then present a summary followed by a detailed description of the results from several studies that discuss the species' response to different noise levels and frequencies (6.2.2, 6.2.3). Since limited information is available for the species cod and herring, international threshold values for fish are also considered which are based on studies of other species. The threshold values are only for injury and mortality, which are assessed to be applicable to fish in general. The difference between the species is expected to be less for physiological damage to internal organs compared to behavioural changes and hearing impairment, which are more strongly linked to the species' sensitivity to both sound frequency and intensity (Popper et al., 2005). However, there are differences in the shape of the swim bladder between species that can affect when damage occurs (Halvorsen et al., 2012b).

### 6.2.1 Hearing of cod and herring

Herring (*Clupea harengus*) is one of the fish species with the highest sensitivity to underwater sound. A gas-filled swim bladder connected to the inner ear, combined with two gas bubbles in the inner ear, contribute to a wide hearing frequency range and low hearing threshold (Doksaeter et al., 2008; Mann et al., 2005; Popper et al., 2004).

The hearing frequency ranges from 30 to 4,000 Hz, and the herring's minimum threshold is 75 dB re 1 μPa at the frequency 100 Hz (Thomsen et al., 2006; Doksaeter et al., 2008) (Figure 38). Because herring have a good ability to perceive sound, the hearing threshold often lies below the background value of the ambient sound. Therefore, their ability to perceive sounds is often limited through masking from the ambient sounds rather than the hearing threshold (Andersson et al., 2011).

Cod do not hear within as wide a frequency range as herring because they do not have the same connection between swim bladder and inner ear. On the other hand, cod have a lower hearing threshold compared to salmon and eel because their swim bladder is closer to their inner ear. Cod hear within the frequency range of 18–470 Hz, and their hearing threshold is lowest at 75 dB re 1 μPa at 160 Hz (Chapman and Hawkins, 1973) (Figure 38).

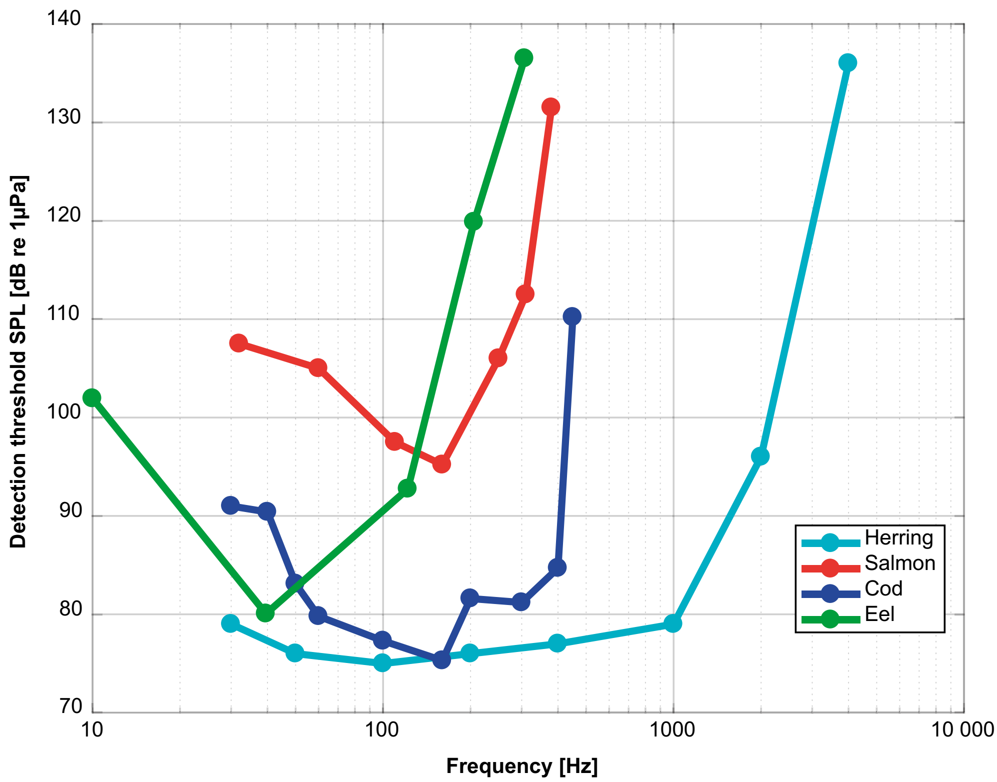



Figure 38. Hearing sensitivity, expressed using the unit sound pressure, in several fish species. Herring (*Clupea harengus*) (Enger 1967), salmon (*Salmo salar*) (Hawkins & Johnston 1978), cod (*Gadus morhua*) (Chapman & Hawkins 1973), eel (*Anguilla anguilla*) (Jerkø et al. 1989) and goldfish (*Carassius auratus*) (Fay 1969). The variation in sensitivity to both the frequency and intensity of sound depends on the anatomical differences between the species. For salmon, and perhaps for eel, the relevant stimulus is most likely particle acceleration rather than sound pressure. Figure from Andersson et al., 2011.

Cod can possibly perceive ultrasound (3 ms 38 kHz pulses), but only at very high volume levels > 194 dB re 1 µPa (Astrup and Møhl 1993, 1998). Cod use sound for communication, for example during aggressive behaviour or spawning, when the males use grunts. The grunts consist of pulses within 30–250 Hz with a very rough estimated source level of between 120 and 133 dB re 1 µPa (Hawkins and Rasmussen 1978; Nordeide and Kjellsby, 1999). Cod also have the ability to position and determine the distance to a sound source, for which both the stimuli from particle acceleration and the sound pressure are important (Schuijf and Hawkins, 1983).

The morphological differences among cod and herring related to swim bladder and hearing mean that studies on these two species can provide an overall picture of how fish generally react to loud, impulsive noise at different frequencies, both in terms of flight behaviour and internal organ damage.

### 6.2.2 Summary of results

The results and references of the studies mentioned in this section are presented in Table 9 as well as in the next section, where each study is described in more detail.

**Table 9. Summary of existing literature on measured noise levels, in which various types of responses have been observed in cod and herring as well as in fish in general. The results are sorted by noise level within each grouping. Note that the noise level is presented both as SPL and SEL, and that there are differences in frequency.**

| | Response | Sound Pressure Level (SPL=dB re 1 μPa/SEL=dB re 1 μPa²s) | | Frequency (Hz) | Reference | Comment | Size (mm) or age (months) |
|---|---|---|---|---|---|---|---|
| Cod | Some mortality (5–10%), internal injury | $SPL_{peak}$ | ~238–242 | Airgun | Booman et al. 1996 | Study on juvenile cod | 100–180 |
| | Internal injury, some recovery | $SPL_{peak}$ | ~230–238 | Airgun | Booman et al. 1996 | Study on juvenile cod | 100–180 |
| | Hearing injury (53%) | $SPL_{peak}$ | 230–242 | Airgun | Booman et al. 1996 | Study on juvenile cod | 100–180 |
| | Fatal injury (90%) | $SPL_{peak}$ | 219–230 | Watergun | Knutsen and Dalen 1985 | Study on juvenile cod | 90–110 |
| | Balance problems, no injury | $SPL_{peak}$ | 214–231 | Airgun | Knutsen and Dalen 1985 | Study on juvenile cod | 90–110 |
| | Fright response, no flight | SPL | 195 | 80–120 (airgun) | Wadle et al. 2001 | Field study, lowest measured sound level | unknown |
| | Injury | SPL | 180 | 50–400 (one tone) | Enger 1981 | Study on cod | 250 |
| | Behavioural change | $SPL_{peak}$ | 140–161 | 150–350 (pile driving) | Mueller-Blenke et al. 2010 | Study on cod | 310–470 |
| | No reaction | SPL | 160 | 470 | Kastelein et al. 2008 | Study on cod | 420–460 |
| | No reaction | SPL | 130 | 200 | Kastelein et al. 2008 | Study on cod | 420–461 |
| | No reaction | SPL | 120 | 100 | Kastelein et al. 2008 | Study on cod | 420–462 |
| | No reaction | SPL | 152–192 | 1,500–6,500 (sonar) | Jørgensen et al. 2005 | Study on juvenile cod | 16–65 |
| Herring | Some mortality (20%) | SPL | 189 | 1,500 (sonar) | Jørgensen et al. 2005 | Study on juvenile cod | 24 |
| | No reaction | $SEL_{cum}$ | 181 | 1,000–7,000 | Doksaeter et al. 2012 | Field study on herring in shoals during feeding migration | unknown |
| | Reaction threshold (50%) | SPL | 160–178 | 4,000 | Kastelein et al. 2008 | Study on cod | 250–300 |
| | No reaction | SPL | 176 | 1,000–7,000 | Doksaeter et al. 2012 | Field study on herring in shoals during feeding migration | unknown |
| | Some mortality (30%) | SPL | 173 | 3,400 (sonar) | Jørgensen et al. 2005 | Study on juvenile cod | 31 |
| | Fright response/Flight behaviour | SPL > | 170 | 1,000–3,000 (sonar) | Jørgensen et al. 2005 | Study on juvenile cod | 24–51 |
| | Reaction threshold (50%) | $SPL_{peak\text{-}peak}$ | 163 | 50–600 | Hawkins et al. 2014 | Field study on European sprat | unknown |
| | Fright response | SPL | 122–138 | 70–200 | Blaxter and Hoss 1981 | Study on cod | 28–170 |
| Cod/Herring | PTS/TTS | SPL ≥ | 205 | | Nedwell et al. 2007 | The value is based on a general value for fish | x |
| | Mild behavourial respons | SPL ≥ | 75–125 | | Nedwell et al. 2007 | The value is based on a general value for fish | x |
| | Strong behavourial response | SPL ≥ | 125–165 | | Nedwell et al. 2007 | The value is based on a general value for fish | x |
| | Strong escape response | SPL ≥ | 165 | | Nedwell et al. 2007 | The value is based on a general value for fish | x |
| International guidance values | Fatal injury | $SEL_{cum}$ | 207 | 100–1,000 (pile driving) | Popper et al. 2014 | Recommended noise level for impact on fish | x |
| | Fatal injury | $SPL_{peak}$ > | 207 | 100–1,000 (pile driving) | Popper et al. 2014 | Recommended noise level for impact on fish | x |
| | Injury with recovery | $SPL_{peak}$ > | 207 | 100–1,000 (pile driving) | Popper et al. 2014 | Recommended noise level for impact on fish | x |
| | Injury with recovery | $SEL_{cum}$ | 203 | 100–1,000 (pile driving) | Popper et al. 2014 | Recommended noise level for impact on fish | x |
| | TTS | $SEL_{cum}$ | 186 | 200,400,1,600 (airgun) | Popper et al. 2014, Popper et al. 2015 | Recommended noise level for impact based on study on carp** | Adult individuals |
| | TTS | $SEL_{cum}$ | 186 | 400 (airgun) | Popper et al. 2014, Popper et al. 2015 | Recommended noise level for impact, adult ind. only | 360–670 |
| Other species | No mortality | $SEL_{cum}$ | 215–222 | 100–1,000 (pile driving) | Debusschere et al. 2014 | Field study on juvenile European sea bass* | 2–4 mos. |
| | Injury (recovery 13 days) | $SEL_{cum}$ | 215 | 100–1,000 (pile driving) | Bolle et al., submitted ms. | Study on juvenile European sea bass* | 104 |
| | Injury recovery | $SE_{Lcum}$ | 217 | 100–1,000 (pile driving) | Casper et al. 2012 | Study on juvenile Chinook salmon** | 99.4 |
| | Injury recovery | $SEL_{cum}$ | 204–213 | 100–1,000 (pile driving) | Casper et al. 2013 | Study on juvenile striped sea bass* | 42; 100 |
| | Injury judged to affect survival (threshold) | $SEL_{cum}$ | 210 | 100–1,000 (pile driving) | Halvorsen et al. 2012a | Study on juvenile Chinook salmon** | 93–115 |
| | Injury judged to affect survival (threshold) | $SEL_{cum}$ | 207 | 100–1,000 (pile driving) | Halvorsen et al. 2012b | Study on cichlids* and juvenile sturgeons** | 84 (6 mos.) & 66 (3–4 mos.), resp. |
| | No significant damage | $SEL_{cum}$ | 205 | 100–1,000 (pile driving) | Bolle et al., submitted manuscript | Study on juvenile European sea bass* | 104 |
| | Injury judged to affect survival (threshold) | $SEL_{cum}$ | 204 | 100–1,000 (pile driving) | Casper et al. 2013 | Study on juvenile striped sea bass* | 42; 100 |
| | Barotrauma on internal organs | $SEL_{cum}$ | 204 | 100–1,000 (pile driving) | Halvorsen et al. 2012b | Study on cichlids* and sturgeons** | 6 and 3–4 mos., resp. |

*Fish with “closed” (physoclistous) swim bladder, including cod. **Fish with “open” (physostomous) swim bladder, including herring.

There is widespread lack of knowledge about injury and mortality in cod and herring during their exposure to high impulsive noise levels. In one study of cod, damaged sensory hair cells were documented at SPL 180 dB re 1 µPa for frequencies between 50 and 400 Hz. The noise that the fish were exposed to lasted a longer time at one frequency ("pure tone") and is therefore difficult to compare with the pile strike. Other studies on juvenile cod that were exposed to the noise of a watergun and an airgun observed increased mortality (90%) at an exposure between SPL 219 and 230 dB re 1 µPa (watergun). Injury to internal organs was observed at between SPL 230 and 242 dB re 1 µPa (airgun). In herring, some mortality has been documented in juvenile individuals at SPL 179 and 189 dB re 1 µPa at frequencies of 3,400 and 1,500 respectively. In the international guidelines, which are based on studies of species other than cod and herring that were exposed to pile driving noise, fatal injuries occur in fish at noise levels of $SEL_{(cum)}$ 207 dB re 1 µPa$^2$s or $SPL_{(peak)}$ > 207 dB re 1 µPa. Injuries that the fish can recover from occur at $SEL_{(cum)}$ 203 dB re 1 µPa$^2$s or $SPL_{(peak)}$ >207 dB re 1 µPa. The studied species have morphological differences in swim bladder shape; as a result, the extent of the injury varies between species and noise level. However, the guidelines are based on the lowest noise levels where damage that is expected to affect the fish's survival has been noted.

It is difficult, if not impossible, to draw general conclusions about behavioural changes in different fish species in their natural environment on the basis of existing studies. This is because there is wide variation in the results of studies addressing behavioural changes in fish exposed to impulsive noise. Because fish hear differently at different frequencies, their behavioural response is also strongly dependent on both sound pressure and frequency, and can vary within species depending on gender, age, fitness, functional stage and the area's importance for fish survival and/or reproduction (Kastelein et al., 2008; Muller-Blenke et al., 2007). In addition, most studies have been conducted in tanks or aquariums. It is important to consider that a fish in captivity can react differently than one that lives in its natural environment.

In cod, a behavioural change has been observed between $SPL_{(peak)}$ 140 to 161 dB re 1 µPa at pile driving noise within the frequency range of 150–350 Hz. A behavioural change need not, however, mean that cod flee an area when exposed to these noise levels. The flight behaviour of cod, on the other hand, has been noted in connection with noise exposure to aiguns. However, this study does not contain any values for the noise levels the fish were exposed to. A fright response has been observed at SPL 195 dB re 1 µPa (80–120 Hz) but the reaction did not result in flight behaviour, indicating that the area could have been important for the survival of the fish. Studies also exist that do not report any behavioural response in cod exposed to noise levels between SPL 120 and 160 dB re 1 µPa at the frequencies 100–470 Hz. All of these values were less than 50 dB above the cod's hearing threshold. Studies of herring have demonstrated that hearing varies between age groups and that juveniles are generally more sensitive than larvae and larger individuals.

The herring's frequency of hearing ranges from 30 to 4,000 Hz, which means that it can register the pile driving noise of all frequencies. In the literature, fright responses and some flight behaviour in herring have been observed between the frequency range of 70–200 Hz and 1,000–3,000 Hz at noise levels of SPL 122–138 and 170 dB re 1 µPa, respectively. The reaction threshold in one study was set at 30 dB above the herring's hearing threshold at 4,000 Hz, as the herring was exposed to approximately SPL 160–178 dB re 1 µPa. During feeding migration, no reaction was noted within 1,000–7,000 Hz at exposure to noise levels of SPL 176 re 1 µPa and SEL 181 re 1 µPa$^2$s. The results indicate that the threshold for escape reactions increases during feeding migration. During a field study on sprat that were exposed to noise levels similar to pile driving noise, a response threshold was observed at $SPL_{(peak\text{-}peak)}$ 163 dB re 1 µPa and $SEL_{(ss)}$ 135 dB re 1 µPa$^2$s within the frequency range of 50–600 Hz.

An important conclusion regarding fishes' flight behaviour as a response to a disturbance is that it does not necessarily affect the fishes at the population level, and that the effect of the impact is strongly linked to the area and time period.

### 6.2.3 Findings from literature review on cod and herring

Findings from the literature review as described in this section are summarised in Table 9. There are just a few studies that have examined injury and mortality in cod exposed to high noise levels. In one of these studies, adult cod are exposed to SPL 180 dB re 1 µPa, which is equivalent to 100–110 dB above the cod's hearing threshold at the cod's most "sensitive" frequencies 150–250 Hz (Enger 1981). The frequencies tested were 50, 100, 200, and varying frequencies between 300 and 400 Hz. The cod were exposed to "pure tones" for 1–5 hours. In the experiment, injury to the inner ear was detected in the form of destroyed hair cells for all tested frequencies, which is expected to affect the fish's hearing and balance systems. The location of the injury in the inner ear was dependent on frequency. It is difficult to draw parallels with pile driving noise or other impulsive noise when the fishes were exposed to a specific frequency for a longer period of time.

Knutsen and Dalen (1985) subjected juvenile cod (110 days, 90–110 mm) with fully developed swim bladders to blasts from an airgun (one small and one large) and a watergun. An airgun and a watergun differ in that the primary pulse from an airgun has a positive pressure while the primary pulse from a watergun has a negative pressure. Booman et al. (1996) state that an airgun causes the same type of injury as a watergun but that the effect occurs at shorter distances for a watergun. Juveniles (110 days) were placed in fine-mesh net cages at a distance of 2–6 metres. In the study, only the sound pressure level (in Pascal) of the primary pulse was presented for the different guns at a 1-metre distance. Roughly translated from the given values, the individuals subjected to the small airgun were exposed to noise levels between SPL 222–205 dB re 1 µPa (from 1–10 m from the sound source), and with the large airgun to noise levels between SPL 231–214 dB re 1 µPa (1–10 m from the sound source). In the treatment with the watergun, the individuals were exposed to noise levels between SPL 230–213 dB re 1 µPa (1–10 m from the sound source). During exposure from the airgun, signs of balance problems were observed but the fish recovered quickly. During exposure to the noise from the watergun at a distance of 2 metres (which at 1 metre had a noise level of SPL 230 dB re 1 µPa and at 5 metres SPL 219 dB re 1 µPa), a mortality rate of 90% was observed. When the fish were dissected, injury was observed in the form of cracked swim bladders and bleeding along the swim bladder and in the liver. At 6 metres (SPL <219 dB re 1 µPa), only balance problems were observed.

Booman et al. (1996) exposed juvenile cod (100–180 mm) to noise from an airgun at a distance of 0.9–1.7 metres from the sound source using noise levels between approximately SPL 242 and 230 dB re 1 µPa. No significant increase in mortality was observed. A few dead individuals (5–10%) were observed only at 0.9 metres from the sound source. However, an autopsy revealed internal organ damage to the swim bladder, kidneys, major veins, hearing organs and eyes of individuals from any distance. Three days after the treatment the injury prevalence was 36%, but after 16 days the incidence of injuries dropped to 6%, indicating a recovery. Using underwater video, it was observed that several individuals had become unconscious. Afterwards, all individuals were transferred to the laboratory for further observation. The treated individuals displayed abnormal swimming behaviour during the first few hours. Both these effects can result in increased mortality in the field.

Mueller-Blenke et al. (2010) studied behavourial response in cod exposed to recorded pile driving noise. The results show a behavioural change in cod between $SPL_{(peak)}$ 140 and 161 dB re 1 µPa. In the experiment, the cod were exposed to a recorded pile driving noise with a maximum noise level of $SPL_{(peak)}$ 170 dB re 1 µPa. The sound levels that were played correspond to a far distance (several km) from a pile driving activity. The results showed variability in individual behaviours in response to the sounds. The pile driving noise in this study was mainly in the range of 150–350 Hz, which falls within the most sensitive frequency range for cod. The reactions consisted of a freezing response, reduced swimming speed while the sound was played, increased swimming speed during the sound exposure and altered swimming direction when the fish were exposed to the sound for the first time.

There are more studies that indicate flight behaviour in cod when exposed to noise from an airgun. A reduced catch probably results from behavioural changes and distribution before and after the sound exposure (Engås et al., 1996; Lokkeborg et al., 2012). In Engås et al. (1996), cod and haddock trawl catches decreased by an average of about 50% while firing was taking place. Decreased catches were observed out to 18 nautical miles (33 km) from the sound source. There was a greater reduction in large cod (>60 cm) than smaller cod. There are no direct values of the fish's sound exposure in these studies that can be used to assess the noise level at which a behavioural response occurs.

Another field study conducted on a coastal reef revealed a C-start response in all reef fish (including cod) exposed to the noise from an airgun. The lowest measured noise level was SPL 195 dB re 1 µPa at a distance of 109 metres from the source (frequency range: 80 to 120 Hz) (Wardle et al., 2001). Interesting to note in this study was that none of the fish moved away from their habitat, perhaps because the area was an important site for the fish's survival.

There are other studies on cod that show no behavioural response (Kastelein et al., 2008; Jørgensen et al., 2005). Some examples of measured noise levels during the study by Kastelein et al. (2008) were SPL 120 dB re 1 µPa at 100 Hz, SPL 130 dB re 1 µPa at 200 Hz and approx. SPL 160 dB re 1 µPa at 470 Hz. All these noise levels are less than 50 dB over the cod hearing threshold (Chapman and Hawkins, 1973). To assess behavioural response, more than 50% of the fish would need to react to the noise. Jørgensen et al. (2005) found no injury or behavioural response in juvenile cod (1.6–6.5 cm) exposed to sonar between the frequencies 1,500–6,500 Hz and with a noise level between SPL 152–192 dB re 1 µPa.

A few studies discuss the behavioural response of herring exposed to impulsive sounds. Kastelein et al. (2008) studied the response of herring (4 individuals) when they were subjected to "pure tones" in the frequency range of 0.1–64 kHz. The study found that a 50% reaction threshold was achieved at 4 kHz (50% of the individuals showed altered swimming behaviour) at 30 dB re 1 µPa over the herring's hearing threshold, which corresponds to the noise level SPL 160–178 dB re 1 µPa. During a field study of sprat (*Sprattus sprattus*), considered to be closely related to herring with similar hearing, a 50% reaction threshold was observed at noise levels similar to pile driving noise of $SPL_{(peak\text{-}peak)}$ 163 dB re 1 µPa and $SEL_{(ss)}$ 135 dB re 1 $µPa^2s$ in the frequency range of 50–600 Hz (Hawkins et al., 2014). Blaxter and Hoss (1981) exposed herring of various sizes to sounds in the frequency range of 70–200 Hz and found a fright response at the noise levels of SPL 122–138 dB re 1 µPa. They determined that the individual's size was crucial for the response, since larvae (28–42 mm) reacted to higher noise levels than larger individuals (140–170 mm); most sensitive were individuals with a length of 80–110 mm. The study by Kastelein et al. (2008) demonstrated the differences between species and that the difference between hearing threshold and reaction threshold varies between different frequencies. According to Blaxter and Hoss (1981), the individual's size also plays an important role.

In studies of the effects of sonar signals on juvenile herring, both mortality (in 2 of the 44 experiments) and behavioural changes (Jørgensen et al., 2005) were noted. At noise levels of SPL 189 dB re 1 µPa at the frequency 1.5 kHz, a 20% mortality rate was observed in medium-sized herring of 2.4 cm; at SPL 179 dB re 1 µPa at the frequency 3.4 kHz, a 30% mortality rate was observed in medium-sized herring of 3.1 cm. No effects of noise exposure were noted on herring around 2 cm since the swim bladder was not yet developed. In behavioural studies, some individuals demonstrated signs of unconsciousness for a few seconds at SPL 176 dB re 1 µPa in the frequency range of 1–3 kHz.

At noise levels in excess of SPL 170 dB re 1 µPa, a fright response as well as certain flight behaviour was observed. Histological studies, however, showed no immediate injury to the organs examined. It was also noted that the herring adapted to noise levels lower than SPL 160 dB re 1 µPa.

Doksaeter et al. (2012) showed that herring, during their feeding migration, did not react to sonar at the frequency 1–7 kHz at noise levels up to SPL 176 dB re 1 µPa and up to SEL 181 dB re 1 µPa$^2$s. Slotte et al. (2004) showed that the presence of herring in an area subjected to seismic measurement was lower than at 20 nautical miles (37 km) away, increasing gradually with distance. Slotte et al. (2004) found that there is a possibility that the migrating individuals might have chosen to take a different direction to avoid the noise.

### 6.2.4 International guidelines and studies of other fish species

The United States has developed guidelines for the effects on fish (mortality and injury) from exposure to impulsive sound (Popper et al., 2014). The guidelines are based on several relatively new laboratory studies on species with different body shapes, swim bladder formation and internal morphology that were exposed to pile driving noise. The swim bladder's formation can affect the fish's sensitivity to sound pressure depending on whether or not the swim bladder has a connection with the esophagus (Simmonds and MacLennan, 2005). A fish is called physostomous if its swim bladder is connected with its esophagus through a thin tube (also called an "open" swim bladder). Through this tube, the fish can expel gas from the swim bladder out through the mouth, thus reducing the negative effect caused by the sound pressure. Fish lacking this connection are called physoclistous ("closed" swim bladder), and they regulate the amount of gas in the swim bladder through secretion and absorption into the blood. This can result in their inability to reduce the volume of gas quickly enough to avoid damage (Halvorsen et al., 2012b). The guidelines are based on the studied species juvenile Chinook salmon (*Oncorhynchus tshawytscha*) (Halvorsen et al., 2011, 2012a; Casper et al., 2012), striped sea bass (*Morone saxatilis*) (Casper et al., 2013) as well as the sturgeon (*Acipenser fulvescens*) and cichlids (*Oreochromis niloticus*) (Halvorsen et al., 2012b). The species were exposed to the same noise levels and exposure time. Like herring, Chinook salmon and sturgeon have an open swim bladder while cod, cichlids and sea bass have a closed swim bladder. The findings showed that cichlids with closed swim bladders sustained more extensive injury than sturgeon and chinook salmon at the highest noise levels ($SEL_{(cum)}$ 216 dB re 1 µPa$^2$s). At lower noise levels ($SEL_{(cum)}$ 204–213 dB re 1 µPa$^2$s), the swim bladder type was of lesser importance as no difference was observed between the species. It is still unclear whether the presence of open or closed swim bladders has any significance in the context of impulsive noise. Popper et al. (2012) conclude that the effects from exposure to pile driving would appear to be consistent across species, whether they are physostomous or physoclistous. Studies in which fish were exposed to sound impulses from underwater explosions showed no difference between physostomous and physoclistous fishes (Yelverton et al., 1975). Hasting and Popper (2005) suggest that fishes with an open swim bladder can be less sensitive to continuous noise, which allows more time to release gas from the swim bladder. Halvorsen et al. (2012b) also point out that a fish's body shape as well as swim bladder placement and size can affect the degree of damage.

In the study of Chinook salmon (Halvorsen et al., 2011; 2012a), the individuals were exposed to various noise levels ($SEL_{(cum)}$ 204–220 dB re 1 µPa$^2$s, $SEL_{(ss)}$ 171–187 dB re 1 µPa$^2$s and number of pile strikes of 960 and 1,920). Based on the results, the study found that the extent of injury depended on both the noise level and number of strikes. In studies of sturgeons and cichlids, the individuals were exposed to noise levels between $SEL_{(cum)}$ 204 and 216 dB re 1 µPa$^2$s and $SEL_{(ss)}$ 174 and 186 dB re 1 µPa$^2$s for 24 minutes, which corresponds to 960 pile strikes (Halvorsen et al., 2012b). Based on the results of the juvenile Chinook study, the authors presented a threshold for injury, which is expected to affect the fish's survival, at $SEL_{(cum)}$ 210 dB re 1 µPa$^2$s. This was reached at an exposure of $SEL_{(ss)}$ 177 dB re 1 µPa$^2$s with 1,920 strikes and at $SEL_{(ss)}$ 180 dB re 1 µPa$^2$s with 960 strikes. Casper et al. (2012) noted an injury recovery in the laboratory for individuals of Chinook salmon exposed to $SEL_{(cum)}$ 217 dB re 1 µPa$^2$s, $SEL_{(ss)}$ 187 dB re 1 µPa$^2$s and 960 strikes. However, it is uncertain whether the fish would recover in their natural environment, where they are likely to have to expend energy on foraging and avoiding predators. At the proposed threshold for juvenile chinook, internal injury in sturgeons and cichlids was still observed; based on the results from these species, the threshold for fatal injury was lowered to $SEL_{(cum)}$ 207 dB re 1 µPa$^2$s, which corresponds to $SEL_{(ss)}$ 177 dB re 1 µPa$^2$s with 960 strikes. Several different types of injury were observed in the studied species. Although the number of injuries and their extent were reduced at the lower noise levels, internal injuries that were judged to be deadly were still observed at the noise level $SEL_{(cum)}$ 204 dB re 1 µPa$^2$s ($SEL_{(ss)}$ 174 dB re 1 µPa$^2$s , 960 strikes). At the lower noise levels, injury to the reproductive organs (gonads) in cichlids was observed, which can decrease reproductive success and thus affect the species at the population level.

Casper et al. (2013) observed internal injuries as well as injury recovery of striped sea bass exposed to noise levels between $SEL_{(cum)}$ 204 and 213 dB re 1 µPa$^2$s ($SEL_{(ss)}$ 171–183 dB re 1 µPa$^2$s). The injuries were more in number and more extensive compared to the other species. At the lower noise level ($SEL_{(cum)}$ 204 dB re 1 µPa$^2$s), the injuries in sea bass were as comprehensive as those of sturgeons and cichlids at the proposed threshold for fatal injury ($SEL_{(cum)}$ 207 dB re 1 µPa$^2$s). The study also observed that the injuries were more extensive in larger individuals (100 mm) compared to juveniles (42 mm). The reason might lie in the difference in the swim bladder's resonance as well as morphological differences, such as swim bladder size and location in relation to other organs and tissues.

Despite some morphological differences among fish species, Popper et al. (2014) assess that the similarities in the results of these studies will allow the guidelines to be applied to fish generally in conjuction with pile driving noise. In the guidelines, fatal injuries occur in fish that detect sound pressure by means of the swim bladder at $SEL_{(cum)}$ 207 dB re 1 μPa²s. Injuries that fish are expected to recover from occur at a noise level of $SEL_{(cum)}$ 203 dB re 1 μPa²s. The guidance values are based on the lowest measured noise levels with recorded injury. Additional studies have found that fish die within a few metres from pile driving (Caltrans, 2004), but there are no data on the noise levels that these fish were exposed to. According to Popper et al. (2014), TTS occurs in fish with swim bladders at $SEL_{(cum)}$ 186 dB 1 μPa²s. The threshold is based on studies on pike (*Esox lucius*), white fish (*Coregonus nasus*) and carp (*Couesius plumbeus*) exposed to airgun noise ($SEL_{(cum)}$ 186 dB 1 μPa²s at various frequencies between 100 and 1,600 Hz) (Popper et al., 2005). Airgun noise is the impulsive noise most similar to pile driving, with the highest energy between 20-50 Hz and decreasing energy at frequencies higher than 200 Hz. TTS was observed in carp at 200, 400 and 1,600 Hz. Like herring, carp have a connection between their swim bladder and inner ear. TTS was also noted in adult pike (length: 360–670 mm) at the frequency 400 Hz. No hearing damage was observed in white fish and juvenile (70–110 mm) pike. All fishes that exhibited TTS in this study recovered within 18 to 24 hours and no internal injury or mortality was observed.

After the U.S. guidelines were presented, additional laboratory studies were carried out on the effects of pile driving noise on juvenile (104 mm) European sea bass (*Dicentrarchus labrax*) (Bolle et al., submitted manuscript, a). The methodology was equivalent to the one used to establish guidance values in Popper et al. (2014). The study noted tissue damage at the exposure of levels at $SEL_{(cum)}$ 215 dB re 1 μPa²s. However, no mortality was observed in the study, and the injured individuals recovered within 13 days from the moment of exposure. No injury was observed at a lower noise level corresponding to $SEL_{(cum)}$ 205 dB re 1 μPa²s.

Debusschere et al. (2014) conducted an *in situ* field study to investigate the effect of pile driving on juvenile sea bass (*Dicentrarchus labrax*) (68 and 115 days, both stages with swim bladder). No increased mortality was observed in the individuals exposed to the pile driving noise between $SEL_{(ss)}$ 181–188 dB re 1 μPa²s with a dominant energy content between 125–200 Hz. In total, throughout the treatment, the individuals were exposed to an $SEL_{(cum)}$ of 215–222 dB re 1 μPa²s. The sound measurements and results support studies done in laboratories (Bolle et al., 2012; Halvorsen et al., 2011; 2012a; 2012b; Casper et al., 2012; 2013), and the author advocates laboratory studies as a suitable approach and alternative to more complicated field studies.

Nedwell et al. (2007) present general guidelines by using the $dB_{ht}$ (species) concept, i.e., a sound at 90 dB re 1 μPa above a species' hearing threshold is presented as 90 $dB_{ht}$. According to these guidelines, TTS generally occurs in fish exposed to 130 $dB_{ht}$, and PTS occurs at repeated exposure. At prolonged exposure (up to 8 hours), the fish can become deaf at noise levels of 90 $dB_{ht}$. According to the guidelines, a mild behavioural response in fish would take place at 0–50 $dB_{ht}$ among a minority of individuals, which probably is not sustained. Sound levels 50–90 $dB_{ht}$ cause a stronger reaction in the majority of the individuals, but acclimation limits the effect. Sound levels at 90 $dB_{ht}$ and higher are expected to produce a strong avoidance reaction in virtually all individuals (see also in Table 3 what $dB_{ht}$ values correspond to in noise levels for cod and herring). Using $dB_{ht}$ is problematic because it has been developed without adequate knowledge of individual species' hearing and disturbance behaviour. The method also does not take into account ambient noise. Assessments made on the basis of the $dB_{ht}$ method must therefore be considered with great caution.

## 6.3 Fish eggs and larvae

All life stages of fish run the risk of being affected by pile driving noise during the construction of offshore wind power. But the earliest life stages – eggs and larvae – are extra sensitive because they are much more fragile and have limited mobility. Fish larvae reactions to pressure changes vary between species and age, as well as on the presence or absence of a swim bladder (Bishai, 1961). The swim bladder is not present in the organism's egg stage but can develop during the larval stage. For benthic fishes such as the common sole (*Solea solea*), the swim bladder is only temporary during the larval stage and regresses after completing metamorphosis at about 25 days after hatching (Bolle et al., submitted manuscript, b). The point when the swim bladder develops during development depends on the species, and for the individuals in the experiment by Bolle et al. (submitted manuscript, b) it ranged between 15–89 days after hatching for three different species.

Today, there are few studies (Table 10) on the effects of pile driving noise on eggs and larvae for cod and herring. Studies on the effects of airgun and underwater explosions, however, can be used to supplement this knowledge gap. Such studies have shown a general increase in mortality of the eggs and larvae of cod and other fish, but only at a very close distance from the sound source, at noise levels around $SPL_{(peak)}$ 242–217 dB re 1 μPa. Calculations made on the basis of the worst possible outcomes from airgun exposure suggest that the expected increase in mortality is low compared with natural mortality, and that the effect on recruitment to the total stock can be viewed as insignificant. Hammar et al. (2014) advocate a possible lower survival rate for cod eggs, larvae and juveniles within a kilometre from the pile driving source. During sensitive periods, such a reduction can lead to direct effects on recruitment, but the authors believe that the effect should not prevent the population's capacity for growth.

**Table 10. Summary of the existing literature on measured noise levels (impulsive sound) and their effects on eggs, larvae and juvenile fish. The results are sorted by sound level within each grouping. Note that the noise level is presented both as SPL and SEL, and that there are differences in frequency. Response described as "increased mortality" is significant.**

| | | Response | Sound Pressure Level ($SPL_{peak}$ re 1 μPa) | | Sound source/ Frequency (Hz) | Author | Reference, sound level | Species | Size (mm) | Age (days after hatching) |
|---|---|---|---|---|---|---|---|---|---|---|
| Eggs | | No effect (some mortality) | $SPL_{peak}$ | 242 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Saithe | x | x |
| Eggs | | No effect | $SPL_{peak}$ | 242 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Cod | x | x |
| Eggs | | No effect (some mortality) | $SPL_{peak}$ | 236 | Airgun | Kostyuchenko 1973 | Turnpenny & Nedwell 1994 | Anchovy | x | x |
| Eggs | | No effect | $SPL_{peak}$ | 222 | Airgun | Knutsen and Dalen 1985 | Davis et al. 1998 | Cod | x | x |
| Eggs | | Increased mortality | $SPL_{peak}$ | 222 | Airgun | Holliday et al. 1987 | Booman et al. 1996 | Anchovy | x | x |
| Eggs | | No effect (some mortality) | $SPL_{peak}$ | 220 | Airgun | Kosheleva 1992 | Turnpenny & Nedwell 1994 | Plaice | x | x |
| Eggs | | No effect | $SPL_{peak}$ | 214 | Airgun | Kosheleva 1992 | Turnpenny & Nedwell 1994 | Plaice | x | x |
| Larvae | Yolk-sac larvae | No effect (some mortality) | $SPL_{peak}$ | 242 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Cod | Unknown | Unknown |
| Larvae | Yolk-sac larvae | Increased mortality | $SPL_{peak}$ | 224 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Turbot | Unknown | Unknown |
| Larvae | Yolk-sac larvae | No effect | $SPL_{peak}$ | 222 | Airgun | Knutsen and Dalen 1985 | Davis et al. 1998 | Cod | Unknown | 1 and 5 |
| Larvae | Yolk-sac larvae | Increased mortality | $SPL_{peak}$ | 220 | Airgun | Holliday et al. 1987 | Davis et al. 1998 | Anchovy | Unknown | 4 |
| Larvae | Yolk-sac larvae | Increased mortality | $SPL_{peak}$ | 217 | Airgun | Holliday et al. 1987 | Davis et al. 1998 | Anchovy | Unknown | 2 |
| Larvae | Yolk-sac larvae | No effect | $SPL_{peak}$ | 210 | 50–1,000, Pile driving (lab) | Bolle et al. 2012 | Bolle et al. 2012 | Common sole | ~ 5.3 | 2 |
| Larvae | Larvae | No effect (retina damage) | $SPL_{peak}$ | 250 (est.) | Airgun | Matishov 1992 | Turnpenny & Nedwell 1994 | Cod | Unknown | 5 |
| Larvae | Larvae | Increased mortality | $SPL_{peak}$ | 223 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Cod | 10–14 | Unknown |
| Larvae | Larvae | No effect (some mortality) | $SPL_{peak}$ | 220 | Airgun | Kosheleva 1992 | Turnpenny & Nedwell 1994 | Plaice | Unknown | Unknown |
| Larvae | Larvae | No effect | $SP_{Lpeak}$ | 217 | 50–1,000, Pile driving (lab) | Bolle et al., submitted ms. | Bolle et al., submitted ms. | Sea-perch | ~ 6 & 14.5 | 18-19 & 38-39 |
| Larvae | Larvae | No effect (damage) | $SPL_{peak}$ | 216 | Airgun | Kostyuchenko 1973 | Davis et al. 1998 | Anchovy | Unknown | Unknown |
| Larvae | Larvae | No effect | $SPL_{peak}$ | 214 | Airgun | Kosheleva 1992 | Turnpenny & Nedwell 1994 | Plaice | Unknown | Unknown |
| Larvae | Larvae | No effect | $SPL_{peak}$ | 210 | 50–1,000, Pile driving (lab) | Bolle et al. 2012 | Bolle et al. 2012 | Common sole | ~ 6.0–7.1 | 8 and 15 |
| Larvae | Post larvae | Increased mortality | $SPL_{peak}$ | 242 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Cod | 19–55 | Unknown |
| Larvae | Post larvae | No effect (some mortality) | $SPL_{peak}$ | ~ 238 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Herring | 24 | Unknown |
| Larvae | Post larvae | Increased mortality | $SPL_{peak}$ | 235-239 | Explosion | Govoni et al. 2008 | Bolle et al. 2012 | Sea bream | 15.9–17.2 | Unknown |
| Larvae | Post larvae | Increased mortality | $SPL_{peak}$ | 229-236 | Explosion | Govoni et al. 2008 | Bolle et al. 2012 | Meagre | 18.0–20.1 | Unknown |
| Larvae | Post larvae | No effect (some mortality) | $SPL_{peak}$ | 235 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Cod | 19–55 | Unknown |
| Larvae | Post larvae | No effect (some mortality) | $SPL_{peak}$ | ~ 230 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Plaice | 17 | Unknown |
| Larvae | Post larvae | No effect (some mortality) | $SP_{Lpeak}$ | ~ 226 | Airgun | Booman et al. 1996 | Booman et al. 1996 | Turbot | 27 | Unknown |
| Larvae | Post larvae | No effect | $SPL_{peak}$ | 222 | Airgun | Knutsen and Dalen 1985 | Davis et al. 1998 | Cod | 20–53 | 56–69 |
| Larvae | Post larvae | No effect | $SPL_{peak}$ | 207 | 50–1,000, Pile driving (lab) | Bolle et al., submitted ms. | Bolle et al., submitted ms. | Herring | 19–40 | 88–89 |

### 6.3.1 Airgun effects

During the 1980s and 1990s, the Norwegian Institute of Marine Research conducted several studies on the effect of airgun noise on eggs, larvae and juveniles. The studies are highly relevant, as they take place in the field and are conducted on fishes including cod and herring.

Knutsen and Dalen (1985) exposed cod eggs, larvae and small juveniles to airgun blasts with a noise level between approx. $SPL_{(peak)}$ 222 and 205 dB re 1 µPa (roughly converted from Pascal at 1 metre from the source of the sound; see explanation in Section 6.2.3.). Eggs, larvae and small juveniles (56–69 days, 20–53 mm) were placed in plastic bags 1–10 metres from the sound source. No mortality or injury was observed in the eggs, larvae or small juveniles.

The damaging effects of airguns on the eggs, larvae and juveniles of several species were also investigated by Booman et al. (1996). Individuals of different life stages were placed at a distance of 0.75–6 metres from the sound source and were exposed to noise levels between $SPL_{(peak)}$ 242–220 dB re 1 µPa. Hatching success and feeding were followed up for individuals from eggs that were treated.

No effects from the noise exposure were observed in eggs from cod and saithe (*Pollachius virens*). Only one group of eggs from saithe exposed to $SPL_{(peak)}$ 242 dB re 1 µPa showed a trend of higher mortality than the other groups. A small but significant increase in mortality was observed in yolk-sac larvae of cod exposed to $SPL_{(peak)}$ 242 dB re 1 µPa. Examinations of these yolk-sac larvae using an optical microscope did not show any tissue damage. No effects from the noise exposure were observed in herring. In other species included in the study, noted was an increased mortality of the yolk-sac larvae of turbot (*Scophthalmus maximus*) that were exposed to $SPL_{(peak)}$ 224 dB re 1 µPa. Clear injury to the free lateral line organs was noted, which can lead to deteriorated fitness in the long run. Other damage that occurred after sound exposure included the formation of vacuoles (blisters) in the brain, spinal cord and eyes. In larvae exposed to $SPL_{(peak)}$ 242 dB re 1 µPa, nerve cells with an abnormally large cell volume were noted. Such an abnormally large increase in volume is probably due to an abnormally strong and rapid pressure change; the authors believe that an injury of this nature in the brain can be considered indirectly fatal.

A significant increase in mortality was observed in cod larvae (10–14 mm, probably without swim bladder) exposed to noise levels at $SPL_{(peak)}$ 223 dB re 1 µPa and in cod post larvae (19–55 mm, probably with swim bladder) exposed to $SPL_{(peak)}$ 242 dB re 1 µPa. For the post larvae of cod, an increased but not significant mortality was observed from exposure to $SPL_{(peak)}$ 235 dB re 1 µPa, 1.5 metres from the sound source. An increased but not significant mortality was also observed in plaice at about $SPL_{(peak)}$ 230 dB re 1 µPa and in turbot at about $SPL_{(peak)}$ 226 dB re 1 µPa. In the exposed post larvae of herring, some increased mortality was observed at about $SPL_{(peak)}$ 238 dB re 1 µPa but no results were significant.

Based on studies like those of Booman et al. (1996), Sætre and Ona (1996) note that although seismic airgun surveys have been demonstrated to cause injury and increased mortality at the individual level, it is still unclear if this has any effect on the actual recruitment of stocks. Without specifying any species, they use the results of Booman et al. (1996) and set an outer limit for 100% mortality at a 2-metre radius from the aigun ($SPL_{(peak)}$ 226 dB re 1 μPa). The estimation is considered a worst-case scenario. According to the authors, the expected daily mortality caused by a seismic survey is so low compared with the natural mortality that the effect on recruitment to the stock can be viewed as insignificant.

### 6.3.2 Explosion effects

Govoni et al. (2008) conducted a field study that examined the effects of shock waves from underwater explosions on the larvae of spot (*Leiostomus xanthurus*, 18.0–20.1 mm, with swim bladder) and pinfish (*Lagodon rhomboides*, 15.9–17.2 mm, with swim bladder). For the exposed spot, mortality increased by 100% at noise levels of $SEL_{(ss)}$ 182–187 dB re 1 $\mu Pa^2s$ and $SPL_{(peak)}$ 229–236 dB re 1 μPa (converted by Bolle et al., 2012). In pinfish that were exposed to the noise levels $SEL_{(ss)}$ 183–186 dB re 1 $\mu Pa^2s$ and $SPL_{(peak)}$ 235–239 dB re 1 μPa (converted by Bolle et al., 2012), mortality increased by 33–100%. The values for $SEL_{(ss)}$ are comparable with pile driving, whereas the level of the measured noise had a higher (peak) value than the noise from pile driving usually has (Bolle et al., 2012). The study indicates that larvae are more susceptible to shock waves from an underwater explosion than larger juveniles and adult individuals (Govoni et al., 2008).

### 6.3.3 Effects of pile driving noise

Bolle et al. (2012) investigated how pile driving noise had an impact on the survival of the larvae of common sole (with swim bladder). A pressure chamber was used, which under controlled laboratory conditions could expose the fish larvae to sound similar to pile driving noise. The method is similar to earlier laboratory studies carried out on fish (with the HICI-FT machine in Halvorsen et al., 2011; 2012a; 2012b; Casper et al., 2012; 2013) and is considered to be equivalent to field studies (Debusschere et al., 2014). The experiment showed no increased mortality during the first 7 days after exposure to noise levels of up to $SEL_{(cum)}$ 206 dB re 1 $\mu Pa^2s$ and $SPL_{(peak)}$ 210 dB re 1 μPa. However, the study focused solely on the lethal effects of pile driving; exposure to such a degree can still result in decreased survival in the long term.

Newer studies with the pressure chamber have been conducted by Bolle et al. (submitted manuscript, b), but with additional larvae of sea bass and herring (both having swim bladders). Several different life stages were tested, but none of the species showed a difference in mortality between the control group and exposed group. The sea bass were exposed to levels of up to $SEL_{(cum)}$ 216 dB re 1 $\mu Pa^2s$ and $SPL_{(peak)}$ 217 dB re 1 μPa, while herring were exposed to noise levels of up to $SEL_{(cum)}$ 212 dB re 1 $\mu Pa^2s$ and $SPL_{(peak)}$

207 dB re 1 µPa. The results were compared with the earlier study in which the larvae of common sole were exposed to noise levels of up to $SEL_{(cum)}$ 206 dB re 1 $µPa^2s$ and $SPL_{(peak)}$ 210 dB re 1 µPa. No change occurred during the 7 days (for the sole) or 10 days (for sea bass and herring) after the exposure. Together, the tested larvae represented the entire range of swim bladder shape types described by Popper et al. (2014). Thus, no distinction between the presence or absence of a swim bladder could be seen, nor between fish with swim bladders connected to the esophagus and those without.

In the latest document, Bolle et al. (submitted manuscript, b) discuss the effects of swim bladder resonance. This is something they suspect does not occur when their chamber is used. Instead, they make use of a theoretical model in which they consider the swim bladder as a gas bubble. From the theoretical study, they could show that the resonance effect was insignificant for swim bladders smaller than 2 mm (radius of the gas bubble). The swim bladders of fish larvae that were tested in the pressure chamber were considerably smaller than that. However, the relevance of the swim bladder's resonance is expected to be higher for bigger swim bladders or at higher levels of high frequencies.

## 6.4 Harbour porpoises

To provide a detailed picture of the effects of noise from pile driving, several studies are considered with specified thresholds for the following responses in harbour porpoises: (1) Displacement of the hearing threshold, known as the threshold shift (TS), (2) Avoidance behaviour and (3) Masking of echolocation ability. The results of the literature review are summarised in Table 11.

### 6.4.1 Hearing of harbour porpoises

The harbour porpoise's echolocation signals are characterised by short, high-frequency clicks (>100 kHz) with a maximum amount of energy around 110–140 kHz (Veerbom and Kastelein, 1995). The hearing ability of harbour porpoises includes frequencies below 1 kHz up to about 140 kHz (Kastelein et al., 2002, 2010) (Figure 39). Their hearing threshold is lowest at frequencies around 100 kHz with thresholds down to SPL 30–40 dB re 1 µPa (Kastelein et al., 2010).

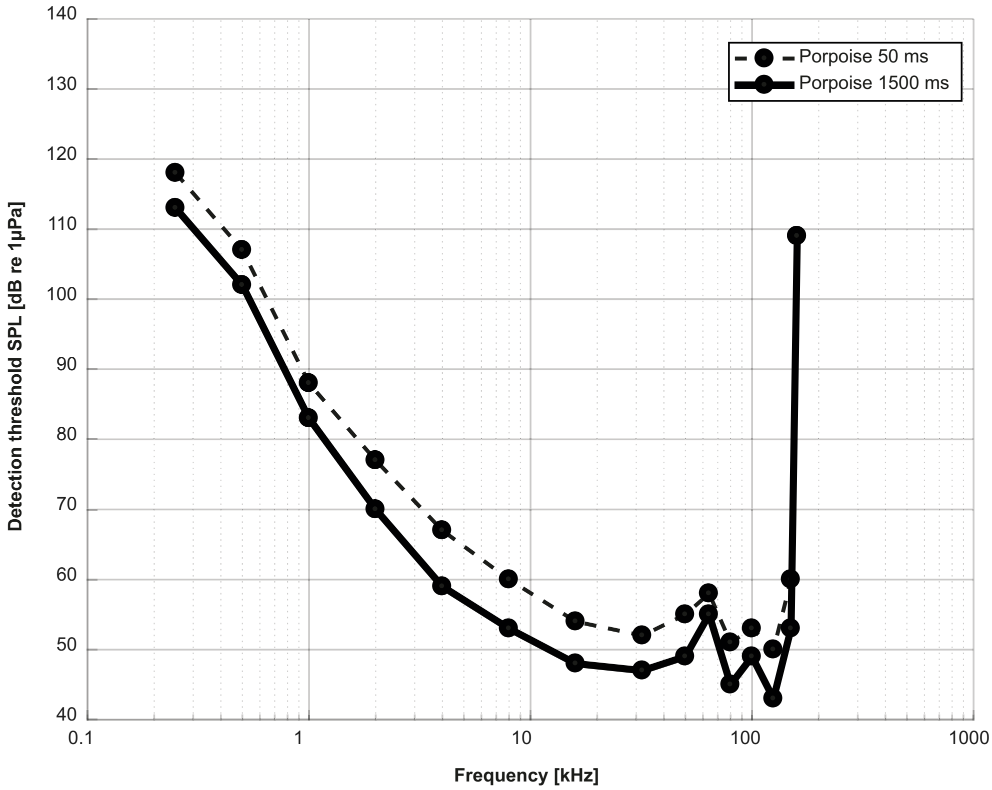



Figure 39. Audiogram for harbour porpoises for two different long tones, 50 ms and 1,500 ms, based on Kastelein et al. (2010).

**Table 11. Summary of existing literature on measured noise levels, in which various types of responses have been observed in harbour porpoises. The results are sorted by sound pressure level within each grouping. Note that the level is presented both as SPL and SEL, and that there are differences in frequency.**

| Response of harbour porpoise | Sound Pressure Level (SPL=dB re 1 µPa/SEL=dB re 1 µPa²s) | | Frequency (kHz) | Reference | Comment |
|---|---|---|---|---|---|
| TTS | $SPL_{peak\text{-}peak}$ | 199.7 | 4 (airgun) | Lucke et al. 2009 | Study on harbour porpoises, Threshold value |
| TTS | $SPL_{peak}$ | 196 | | NOAA 2015 | Threshold value, based on high-frequency cetaceans |
| TTS | SPL | 160 | 45 | Popov et al. 2011 | Study on a related speciesof harbour porpoise, One sound level |
| TTS | SEL | 188–196 | 1–2 | Kastelein et al. 2014 | Study on harbour porpoises, Threshold value |
| TTS | SEL | 190 | 1.5 | Kastelein et al. 2013 | Study on harbour porpoises, One sound level |
| TTS | $SEL_{cum}$ | 183 | 45 | Popov et al. 2011 | Study on a related speciesof harbour porpoise, One sound level |
| TTS | $SEL_{cum}$ | 180 | 4 and 8 (pile driving) | Kastelein et al. 2015 | Study on harbour porpoises |
| TTS | SEL | 163–172 | 4 | Kastelein et al. 2012 | Study on harbour porpoises, Threshold value |
| TTS | $SEL_{ss}$ | 164 | 4 (airgun) | Lucke et al. 2009 | Study on harbour porpoises, Threshold value |
| TTS | $SEL_{cum}$ | 162 | | NOAA 2015 | Threshold value, unweighted*, based on high-frequency cetaceans |
| TTS | $SEL_{ss}$ | 146 | 4 and 8 (pile driving) | Kastelein et al. 2015 | Study on harbour porpoises |
| TTS | $SEL_{cum}$ | 139 | | NOAA 2015 | Threshold value, weighted*, based on high-frequency cetaceans |
| PTS | $SPL_{peak}$ | 230 | | Southall et al. 2007 | Threshold, unweighted*, based on other species of marine mammals |
| PTS | $SPL_{peak}$ | 202 | | NOAA 2015 | Threshold value, based on high-frequency cetaceans |
| PTS | SEL | 198 | | Southall et al. 2007 | Threshold, weighted*, based on other species of marine mammals |
| PTS | SEL | 183 | 45 | Popov et al. 2011 | Study on a related speciesof harbour porpoise, One sound level |
| PTS | $SEL_{cum}$ | 177 | | NOAA 2015 | Threshold value, unweighted*, based on high-frequency cetaceans |
| PTS | $SEL_{cum}$ | 154 | | NOAA 2015 | Threshold value, weighted*, based on high-frequency cetaceans |
| Negative behavioural reaction | $SPL_{peak\text{-}peak}$ > | 174 | 4 (airgun) | Lucke et al. 2009 | Study on harbour porpoises |
| Negative behavioural reaction | SEL | 145 | 4 (airgun) | Lucke et al. 2009 | Study on harbour porpoises |
| Avoidance behaviour | SEL50 | 144–146 | 0.1–1 (pile driving) | Schubert 2015 | Study on harbour porpoises, threshold |
| Avoidance behaviour | SEL | 139–145 | 0.1–1 (pile driving) | Dähne et al. 2013 | Study on harbour porpoises, threshold |

*Frequency weighting is a method of quantitatively compensating for the difference in hearing ability between different frequencies for a sound (see also Section 2.3.2)

### 6.4.2 Threshold shift (TS)

Exposure to extremely high noise levels can cause permanent hearing damage known as permanent threshold shift (PTS) or a temporary hearing loss known as temporary threshold shift (TTS) of 6 dB in the frequency range where the acoustic energy is found. According to Southall et al. (2007), the difference between these two effects is that PTS is considered physiological damage while TTS is only auditory fatigue, an effect that is reversible. But the body of knowledge on this subject has grown since the Southall et al. (2007) study, and the perception of TTS and PTS is different today. Tougaard et al. (2015) believe that it is unclear whether TTS is considered to be a physiological injury in marine mammals. The effects on the hearing organ differ between TTS and PTS. PTS involves damage to the sensory cells in the hearing organ, while TTS partly occurs due to swelling of specific nerve endings in the hearing organ. The authors refer to the experiments conducted on terrestrial mammals, which demonstrated that hearing does not necessarily recover fully after a powerful TTS. Tougaard et al. (2015) also discuss that an animal exposed to repeated and severe TTS is likely to develop a form of PTS.

According to Southall et al. (2007), the onset of PTS takes place during exposure to sound pressure levels that generate a TTS corresponding to a reduction in the hearing threshold of 40 dB. The definition of the criteria is based on the knowledge about the auditory system anatomy of marine and terrestrial mammals and from extrapolation of TTS data from two other species of marine mammals: bottlenose dolphins (*Tursiops truncatus*) and beluga (*Delphinapterus leucas*). The threshold for PTS is to be expected for sound pressurelevels around $SPL_{(peak)}$ 230 dB re 1 µPa (unweighted) or at SEL 198 dB re 1 µPa$^2$s (M-weighted) (Southall et al., 2007). Frequency weighting is a method of quantitatively compensating for the difference in hearing ability at different frequencies for a sound (Southall et al., 2007). M-weighting is used for marine mammals; see Section 2.3.2.

A modified version of the M-weighting (Southall et al., 2007) has been developed by Finneran and Jenkins (2012) by supplementing the function with new data containing frequencies with an increased sensitivity to sound-induced auditory threshold shift. The function has also been extrapolated and applied to whales in the functional group high-frequency cetaceans – this includes the harbour porpoise – by the National Oceanic and Atmospheric Administration (NOAA, 2015). NOAA has since produced guidance on assessing the effects of anthropogenic sound on marine mammals through PTS and TTS threshold values. The guidelines provide a dual threshold in which the noise level that is first exceeded should apply. The PTS thresholds are defined as $SPL_{(peak)}$ 202 dB re 1 µPa and $SEL_{(cum)}$ 154 dB re 1 µPa$^2$s (weighted), and $SPL_{(peak)}$ 202 dB re 1 µPa and $SEL_{(cum)}$ 177 dB re 1 µPa$^2$s (unweighted). Notably, because the criteria for PTS from impulsive noise in Southall et al. (2007) and NOAA (2015) are not based on empirical data for the harbour porpoise, the result should only be interpreted as an indication and not an absolute measure.

In a study by Popov et al. (2011) a related species of the harbour porpoise, the Yangtze finless porpoise (*Neophocaena phocaenoides asiaeorientalis*), was exposed to 3-minute pulses of a 0.5-octave frequency range noise of around 45 kHz at SPL 160 dB re 1 µPa. Converted to SEL gives the equivalent of 183 dB re 1 uPa$^2$s (Tougaard et al., 2015). The noise caused a TTS of 45 dB that was so strong that it was categorised as PTS. But this sound differs from pile driving noise, which has pauses of silence between strikes, which the study by Popov et al. (2011) does not have. It is possible that the animal has the time to recover slightly from TTS during these pauses of silence. The hearing ability of harbour porpoises and the related species exhibit no major differences according to Popov et al. (2006), a result that is interpreted as meaning that the two species have generally equivalent hearing (Tougaard et al., 2015). The result should therefore be representative even for harbour porpoises. More recent studies have shown a clear frequency dependence in marine mammals (Finneran et al., 2015; Kastelein et al., 2015). In addition, the hearing of harbour porpoises at 45 kHz is significantly better compared with frequencies below 1 kHz, where the pile driving noise has the most energy; most likely, the demonstrated threshold is an underestimation of the actual value that induces PTS from pile driving noise. Against this background, the noise level at 183dB re 1 µPa$^2$s SEL, which has been recommended as the PTS threshold, should be regarded with caution.

NOAA (2015) defined thresholds for TTS onset as $SPL_{(peak)}$ 196 dB re 1 µPa and $SEL_{(cum)}$ 139 dB re 1 µPa$^2$s (weighted), and $SPL_{(peak)}$ 196 dB re 1 µPa and $SEL_{(cum)}$ 162 dB re 1 µPa$^2$s (unweighted). According to Southall et al. (2007), TTS onset is defined as an elevation in the hearing threshold by 6 dB. Below, we summarise some studies used to establish TTS thresholds for harbour porpoises.

In an experiment by Lucke et al. (2009), harbour porpoises were exposed to airgun blasts in order to obtain data on TTS induced by single pulses. The nature of the noise from an airgun is similar to pile driving noise and therefore applicable in this study. The results showed that at 4 kHz, TTS exceeded the predefined criteria for the sound pressure level $SPL_{(peak\text{-}peak)}$ 200 dB re 1 µPa and a sound exposure level of $SEL_{(ss)}$ 164 dB re 1 µPa$^2$s.

Kastelein et al. (2012) subjected harbour porpoises to an octave-band white noise centered at 4 kHz over different time lengths (from 7.5 minutes up to and including 4 hours). The thresholds for TTS were noted at SEL 163–172 dB re 1 µPa$^2$s. Kastelein et al. (2013) induced TTS equivalent to 14 dB in harbour porpoises following exposure to a long and continuous 1.5 kHz tone at SEL 190 dB re 1 µPa$^2$s. Since only one noise level was applied, no threshold could be deduced from the experiment. In another study by Kastelein et al. (2014), TTS thresholds in harbour porpoises following exposure to sounds between 1–2 kHz of different characters (noise levels between SPL 144–179 dB re 1 µPa and exposure time between 1.9–240 minutes) (SEL175 and 205 dB re 1 µPa$^2$s according to Tougaard et al. 2015). In the study, TTS thresholds could be recorded within the levels of between SEL

188 and 196 dB re 1 µPa²s. Furthermore, in a study by Kastelein et al. (2015) harbour porpoises were exposed to recorded pile driving noise with an unweighted sound exposure level of $SEL_{(ss)}$ 146 dB re 1 µPa²s ($SEL_{(cum)}$ 180 dB re 1 µPa²s ) for 60 minutes. A small but significant TTS was noted only at frequencies of 4 and 8 kHz, respectively, despite the fact that the majority (86%) of the acoustic energy was in the frequency range of 500–800 Hz. The hearing ability for frequencies that are significant for echolocation 125 kHz (±10 kHz) was not affected by exposure to the pile driving noise.

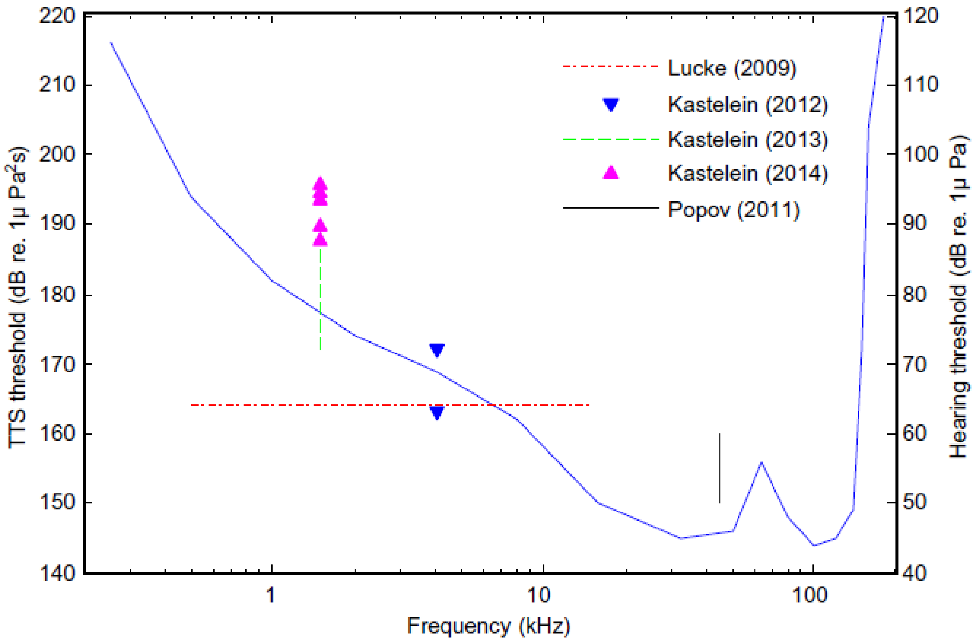



Figure 40. Sound exposure levels (left scale in SEL) required to induce TTS at 6 dB according to stated references in the figure and summarised in the text. The blue line is an audiogram with the scale to the right. Estimated threshold ranges are given only for data from Popov et al. (2011) and Kastelein et al. (2013). Stimulation in Lucke et al. (2009) consisted of pulses from an airgun; therefore, the frequencies are stated as an interval. From Tougaard et al., 2015.

According to (Figure 40), from Tougaard et al. (2015), based on limited information about TTS from harbour porpoises (3 studies) and a related species of harbour porpoises (1 study), a preliminary limit is proposed for TTS onset in harbour porpoises at noise levels of 100–110 dB above the porpoises' hearing threshold for a pure tone at a specific frequency.

### 6.4.3 Behavourial response

In the study by Lucke et al. (2009) in which harbour porpoises were exposed to airgun blasts, the animals exhibited consistently negative behavioural reactions at sound pressure levels over $SPL_{(peak-peak)}$ 174 dB re 1 µPa or SEL 145 dB re 1 µPa²s.

During construction of the Alpha Ventus wind farm, avoidance in the harbour porpoises was noted on the order of 20 km. The noise levels at

25 km were calculated to be equivalent of SEL 139–145 dB re 1 µPa in the study (Dähne et al., 2013). In a similar study, during construction of the Borkum West II wind farm in the North Sea the porpoises exhibited avoidance behaviour at noise levels down to $SEL_{50}$ 144–146 µPa²s ($SEL_{50}$ = mean, see Section 3.2.3). At lower noise levels, no avoidance behaviour was noted (Diederichs, 2014; Schubert et al., 2015).

In the consolidated works of Southall et al. (2007) regarding marine mammals and their reactions to noise, no threshold levels for avoidance in harbour porpoises are stated with regard to pile driving noise, due to the absence of empirical data. In 2015, however, Tougaard et al. (2015) publishd an article that discussed how noise influences whales, with a focus on harbour porpoises. The article highlighted the effects of pile driving noise on harbour porpoises and the levels at which their avoidance behaviour is to be expected. The authors highlight problems in establishing thresholds since results from different studies are compared.

Because the duration of a pulse affects audibility in a harbour porpoise, the authors recommend converting measured values using Leq-fast, which is the use of a fixed time constant (0.125 ms) and dB re 1 µPa (RMS, see fact box 1). In this way, results using studies with short sounds but different durations can be compared. For more information about the conversion and underlying theory, the reader is referred to Tougaard et al. (2015). The article illustrates Leq-fast using a figure that converts and compares the values for pile driving operations and audiograms for porpoises (Figure 41). Based on the results in this figure, Tougaard et al. (2015) describe that the thresholds for avoidance in harbour porpoises lies within the range of 40–50 dB above the hearing threshold; however, the ambient noise must be taken into account before the values are used.

Fact box 1. RMS refers to the average sound pressure level over a given unit of time. Tougaard et al. (2015) propose converting the average sound pressure level using a given time constant (0.12 s) according to the following: Leq-fast = $L_{eq} + 10\log(1-e^{-d/t})$, where *d* is the duration of the sound pulse in seconds and *t* a time constant (0.125 s). The time constant is based on knowledge of human hearing (integration time for sound). The result enables a direct comparison of thresholds for various signals of varying duration. The conversion is recommended when discussing thresholds for behavioural response.

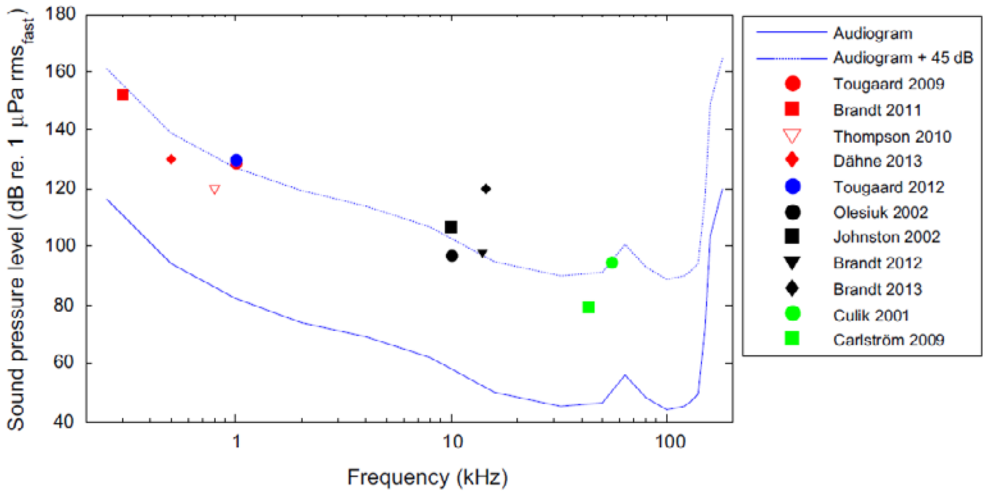



Figure 41. Estimated thresholds for behavioural reactions (negative phonotaxi) from several field studies covering: pile driving (red & blue), acoustic seal deterrents (black) and harbour porpoise pingers (green). For acoustic seal deterrents and harbour porpoise pingers, the x-axis corresponds to the frequency that was most likely audible to the porpoises. For pile driving, the x-axis represents the frequency that contained the loudest noise. All estimated thresholds were converted to Leq-fast. Open symbols indicates a study in which no reaction was observed at the specified sound pressure level. The solid line represents the harbour porpoise's audiogram according to Kastelein et al. (2010) and the dashed line corresponds to the audiogram +45 dB. From Tougaard et al., 2015.

### 6.4.4 Masking of echolocation ability

The character of a background sound with respect to sound pressure level and frequency range affects its audibility. Masking occurs when a sound (like ambient noise) interferes with the detection of another sound (like a signal). The degree of disturbance or masking is influenced by the level and the difference in frequencies between background sound and signal. Masking is most efficient when the two sounds have the same frequency range (Kastelein and Jennings, 2012). A pure tone is masked mainly by sound from nearby frequencies in a critical range of frequency. Sounds from frequencies outside this band have only a small influence on the detection of a signal, as long as the masking sound is not very powerful (Kastelein and Jennings, 2012). It is possible to compromise the ability of harbour porpoises to echolocate by increasing the power of the ambient noise within the frequency range that matches the animal's echolocation click. According to Kastelein & Jennings (2012), however, it is unlikely that this occurs upon exposure to pile driving noise because harbour porpoises produce clicks within a narrow high-frequency band with a maximum sound energy at 110–140 kHz (Veerboom and Kastelein, 1995), and the nature of a pile driving noise has most of the energy at frequencies below 1 kHz. This is also the reasoning of Andersson and Johansson (2013), who say that it is unlikely that the harbour porpoise's echolocation is masked to any greater degree due to sonar from military vessels since the frequencies between sounds do not overlap. This reasoning is also supported by Kastelein et al. (2015) – that is, TTS caused by pile driving noise is assumed to have little or no effect on the harbour porpoise's echolocation ability but can influence its perception of its surroundings.

# 7 International guidelines

## 7.1 Summary

Guidance or threshold values for regulating underwater noise during construction have been developed by several different countries and international organisations. These efforts began in the late 1990s with seismic surveys, and have in recent years been applied to impact pile driving (Wier and Dolman, 2007). Most of the guidelines focus on marine mammals and do not address fish or eggs and larvae. Only one study currently discusses the thresholds for fish (Popper et al., 2014), in Section 6.2 (Cod and herring) and Chapter 2 (Recommendations on thresholds for pile driving noise).

Early guidance contains no set thresholds but instead addresses more of the visual and technical methods for reducing the impact. In recent years, countries such as Denmark, the Netherlands, Germany and the United States have proposed thresholds for impulsive sounds like pile driving noise (Table 12). Sounds that exceed these thresholds are expected to affect marine mammals in the form of behavioural changes and altered hearing (TTS, PTS).

**Table 12. List of national thresholds for effects on marine mammals. All values are unweighted.**

| Country | Thresholds | |
|---|---|---|
| Denmark | Threshold for single pulses:<br>TTS at $SEL_{ss}$ 164 dB re 1 $\mu Pa^2s$<br>PTS at $SE_{Lss}$ 179 dB re 1 $\mu Pa^2s$ | Threshold for a series of pulses (≥1 h):<br>TTS at $SEL_{cum}$ 175 dB re 1 $\mu Pa^2s$<br>PTS at $SEL_{cum}$ 190 dB re 1 $\mu Pa^2s$ |
| United States | Threshold for impulsive sounds:<br>TTS at $SPL_{peak}$ 196 dB re 1 μPa or $SEL_{cum}$ 162 dB re 1 $\mu P^2s$<br>PTS at $SPL_{peak}$ 202 dB re 1 μPa or $SEL_{cum}$ 177 dB re 1 $\mu P^2s$ | |
| Germany | Threshold for disturbance:<br>$SEL_{ss}$ 140 dB re 1 $\mu Pa^2s$ | Threshold at 750 m:<br>SEL 160 dB re 1 $\mu Pa^2s$ or<br>$SPL_{peak\text{-}peak}$ 190 dB re 1 μPa |
| Netherlands | Threshold for disturbance:<br>$SEL_{ss}$ 140 dB re 1 $\mu Pa^2s$ | Threshold at 750 m (Borssele project):<br>SEL 159 dB re 1 $\mu Pa^2s$ (lowest)<br>SEL 172 dB re 1 $\mu Pa^2s$ (highest) |

## 7.2 Guidelines

Four agencies within the European Union (EU) have produced guidelines on the effects of underwater noise (ACCOBAMS, ASCOBANS, OSPAR and ICES). ACCOBAMS, together with ASCOBANS, formed a working group that developed a literature review in which they describe the political and technical efforts implemented to date (ACCOBAMS, 2013).

The document provides a summary of the agencies' proposed guidelines as regards the general concept and specific action plans for offshore construction, with the goal of reducing the negative effects of noise. The recommendations aim to ensure that the following goals are met: accurately estimate risks through the use of acoustic models; plan activities in low-risk areas and avoid high-risk areas; use ramp-up to give animals the chance to escape from an exposed area; monitor relevant areas visually and acoustically; and use the latest technology to minimise noise levels.

### 7.2.1 Great Britain

In 2010, the British government published a document in co-operation with the Joint Nature Conservation Committee (JNCC) that presents a protocol for how to limit the potential effects of pile driving (JNCC, 2010). The protocol does not address measures for the mitigation of disturbances, but is rather designed to reduce the risk of injury or fatality of marine mammals in the immediate vicinity of pile driving operations. Much of the content is based on JNCC's earlier report on the effects of seismic activity on marine mammals (JNCC, 1998). They believe that the noise levels at a seismic survey can be equal to those associated with pile driving, and that it is appropriate to adopt similar alleviation measures.

The protocol provides information about the execution of project planning, the role of Marine Mammal Observers (MMO), the use of passive acoustic monitoring (PAM) during execution, and communication between the crew and the people responsible for mitigation measures. Around the pile driving site a mitigation zone should be established, with at least a 500-metre radius from the sound source. It is within this area that PAM and MMOs should monitor for the presence of marine mammals before pile driving begins. This zone represents the area within which a marine mammal can be injured. In addition to recommendations for ramp-up and possible use of acoustic deterrent devices (ADDs), there are no guidelines on the recommended noise level limits.

Earlier versions of JNCC's guidelines for seismic surveys have been criticised for lacking a scientific basis and demonstrable effects of the proposed alleviation methods (Parsons et al., 2009; Wright and Cosentino, 2015). The latest edition was produced before this criticism was revealed, so the standing guidelines still contain the criticised shortcomings. During the planning phase, operators should investigate the possible presence of marine mammals within the affected area based on known data, which Parsons et al. (2009) believe is not possible considering the limited information available. A major flaw lies in the assumption that an individual in the mitigation zone will swim away from the source during the disturbance, something that may seem to be common sense but that the authors do not believe is scientifically sound. The complex sound image generated in the water could lead the animal to swim toward the source instead, where it could experience a concentration of higher noise at greater distances. An additional common-sense technique that JCNN recommends is using ramp-up, a frequently used method for the alleviation of impulsive sounds but one that critics claim has not been studied fully. Avoidance can also take the form of vertical movement instead of horizontal and, as a result, the individual remains in the area at the start of full-scale pile driving (Parsons et al., 2009). Field studies on the effects of ramp-up have only just begun, and according to Wright and Cosentino (2015) the results are too few and preliminary to draw any firm conclusions. Simulations of ramp-up's effectiveness exist but are simplifications based on groundless assumptions, according to critics. They also believe that the established distances lack a scientific basis.

One of the main problems with using visual methods to detect marine mammals is that several species live in deep waters and rarely can be seen at the surface. On the few occasions that an animal actually appears at the surface, the success of detection is also affected by the surroundings and the MMO observer. Weather conditions and time of day can significantly complicate detection, while the training and experience of the observer affects their ability to detect individuals (Parsons et al., 2009; Wright and Cosentino, 2015).

Overall, Parsons et al. (2009) consider that JCNN's guidelines need to be updated to better reflect the current lack of knowledge. According to Parsons et al. (2009), the best way to alleviate the effects of impulsive sound on marine mammals is simply to avoid the animals in time or space. For information about other proposed guidelines and methods for alleviating the effects of impulsive noise, read the conclusions of Parsons et al. (2009).

### 7.2.2 Overall picture from other countries

In 2007, Wier and Dolman published a literature review in which they reviewed the different types and the effectiveness of the mitigation measures used for industrial seismic studies around the world. The UK's JNCC was the first regulatory body that issued regulations for the alleviation of the effects of seismic surveys on marine mammals. Since 1998 the regulations have been updated, and the latest version from 2010 is the one used as the basis for their pile driving guidelines. Many of the guidelines produced by other countries are more or less based on this first document and, as a result, contain many of the shortcomings that have been criticised by Parsons et al. (2009).

The countries and areas addressed by Wier and Dolman (2007) include Alaska, Australia, Brazil, California, Canada, New Zealand, the Gulf of Mexico and Russian Sakhalin. Overall, three primary measures are used: the implementation of operational techniques like ramp-up, the detection of animals in the vicinity of the source (within an mitigation zone) and implementation of an active response (pausing the operation), as well as scheduling surveys during periods and in areas where marine mammals are not present.

Many details differ between the countries, including the size of the relief zone, which species the guidelines apply to, and the use of pauses and interruptions. However, altogether many of these methods lack a scientific basis.

Wier and Dolman (2007) provide recommendations for a standardised set of global guidelines on the basis of the reviewed works. As for their recommended mitigation zone, they believe that although it is generally assumed that marine mammals will likely be harmed by received levels of 180 dB re 1 $\mu Pa_{(RMS)}$, the value is not sufficiently cautious. They write that behavioural changes have occurred at received levels of a minimum of 160 dB re 1 µPa(RMS) and that the mitigation zone should be calculated on the basis of this value, with reservations for future research within the field. The distance for the mitigation zone should be calculated for each specific location with variables measured in the relevant area and verified in the field at the start of the installation.

### 7.2.3 Ireland

In August 2007, the Irish Department of the Environment, Community and Local Government developed a Code of Practice for the Protection of Marine Mammals during Acoustic Bottom Surveys in Irish Waters. In January 2014, a document was created to review and develop the previous action policies (Department of Art, Heritage and the Gaeltacht, 2014). This document describes the official guidelines and code of practice under Regulation 71 of the European Communities (Birds and Natural Habitats) Regulations 2011 (S.I. No. 477 of 2011).

Under Irish legislation, disturbance and injury to marine mammals from introduced anthropogenic noise are considered a crime. Injury includes temporary and permanent tissue damage as well as TTS. Anthropogenic noise that has the ability to cause TTS in marine mammals is regarded as a potential source of both disturbance and injury. The last official document intends to set the general framework for action and provide guidance when planning and assessing specific noise-producing activities, such as pile driving, in Irish waters.

Before a pile driving operation begins, the area within 1,000 metres from the sound source must be monitored (MMO), and the pile driving must not begin until the area is free from marine mammals. For operations where the noise exceeds $SPL_{(peak)}$ 170 dB re 1Pa at 1 m, ramp-up techniques must be used in which the sound energy or transmission frequency is increased gradually. In addition to risk characterisation and management, no actual thresholds are established for permissible noise levels during pile driving. The noise levels addressed during the discussion on thresholds for TTS and PTS come from Southall et al. (2007), but caution is urged when using these in light of recent research (e.g., Lucke et al. 2009).

## 7.3 Threshold values

### 7.3.1 Denmark

In Denmark, energinet.dk was formed in June 2014. It is a working group with the mandate to examine how underwater noise from pile driving can be regulated in order to take due consideration of marine mammals. The group's findings and recommendations are presented in a technical report (Skjellerup et al., 2015). Denmark's final rules on noise regulation are still under review; in 2016, a revision of the recommendations was published containing updates from the most current research (Tougaard, 2015).

The technical report points out that the use of SEL is now widely accepted as a measure for TTS (Table 13). SEL takes into account the duration of the sound, while $SPL_{(peak)}$ does not. New studies have indicated a frequency dependence in marine mammals in relation to temporary and permanent hearing injury (Finneran, 2015; Kastelein et al., 2015). Regarding TTS from single pulses for harbour porpoises, Lucke et al. (2009) consider them to be most representative for the effects of pile driving noise. Lucke et al. (2009) investigated TTS upon exposure to individual pulses from an airgun and obtained a threshold of $SEL_{(ss)}$ 164 dB re 1 μPa²s. On this basis, they calculate a PTS threshold at single pulses of $SEL_{(ss)}$ 179 dB re 1 μPa²s (TTS + 15 dB). The results from Kastelein et al. (2015) are not included in the calculation of thresholds for single pulses because they consider that method uncertainties exist. The greatest uncertainty lies in the sound level that the exposed harbour porpoises were in fact subjected to during the experiment. They have instead chosen to calculate a more conservative threshold on the basis of the lower quartile of measured noise levels in the study, giving a threshold for TTS from repeated pile driving pulses of $SEL_{(cum)}$ 175 dB re 1 μPa²s and a PTS of $SEL_{(cum)}$ 190 dB re 1 μPa²s. These values apply only for exposure to a long pile driving series (≥ 1 hour), but can be extrapolated and used for longer exposures.

**Table 13. Thresholds for TTS and PTS for harbour porpoises, from Skjellerup et al. (2015) and Tougaard (2015).**

| Harbour porpoises | | |
|---|---|---|
| | **Single pulse** | **Series of pulses (≥ 1 h)** |
| TTS (dB re. 1 μPa²s) | $SEL_{ss}$ 164 | $SEL_{ss}$ 179 |
| PTS (dB re. 1 μPa²s) | $SEL_{cum}$ 175 | $SEL_{cum}$ 190 |

Sounds that fall below the PTS and TTS thresholds can still lead to changes in the behaviour of single individuals. If enough individuals are affected, this can have negative consequences for the entire population. Skjellerup et al. (2015) discuss the thresholds for the management and conservation of entire populations, but believe that knowledge is too flawed concerning how direct, short-term changes in behaviour can be translated into effects on an entire population. Several studies have examined the behavioural response of harbour porpoises exposed to noise from pile driving. They believe that the most reliable study is from Dähne et al. (2013), which indicates a flight threshold value for harbour porpoises of $SEL_{(ss)}$ 140 dB re 1 μPa²s.

The research group in Denmark (Skjellerup et al., 2015) believes that deliberate harm to marine mammals, such as PTS, is not acceptable and that appropriate measures should be taken to avoid exposure to noise above the PTS threshold. The documents do not address how these measures should be put into practice.

### 7.3.2 Germany

Since 2011, the German government has invested in the development of the country's electricity supply for switching to renewable alternatives. By 2050, the bulk of the country's energy supplies are to consist of renewable energy, with wind power as one of the cornerstones. Germany's Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU) has created guidelines for how to protect the harbour porpoise from harmful effects during the construction of offshore wind farms in the German exclusive economic zone in the North Sea (BMUB, 2014). The underwater noise generated by pile driving operations at offshore wind farms can have significant adverse effects on marine mammals, both on the individual and the population level.

The guidelines recommend utilising the best available technology to minimise noise exposure and other adverse effects on the marine environment. The German Federal Maritime and Hydrographic Agency (BSH) has established a dual threshold for permissible noise levels, which must not be exceeded at 750 metres away from the source, of SEL 160 dB re 1 µPa$^2$s or $SPL_{(peak\text{-}peak)}$ 190 dB re 1 µPa. With the set guidelines, disturbances are expected within a radius of 8 kilometres around the source. At this distance, the calculated noise levels are expected to decline from SEL 160 dB re 1 µPa$^2$s to SEL 140 dB re 1 µPa$^2$s, which are thresholds for disturbance that cause avoidance and flight. In areas where noise levels are above the threshold value, intrusive methods such as acoustic deterrent devices should be used to minimise the risk of injury to the animals. For more information about how the measures should be implemented in practice, read the German standards document (BSH, 2013).

### 7.3.3 United States

On 23 July, 2015, the U.S. scientific agency NOAA (National Oceanic and Atmospheric Administration) published its third version of the draft "Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing" (NOAA, 2015). They set acoustic threshold levels for exposure to impulsive anthropogenic noise levels above which marine mammals are expected to experience TTS or PTS. The document has undergone internal and external peer reviews, and is at the time of writing in the final stages of public consultation. Minor adjustments to the guidelines might be made before final publication.

The hearing of marine mammals differs between species in terms of sensitivity and frequencies. To reflect this variable hearing ability, NOAA uses recommendations from Southall et al. (2007) and divides marine mammals into functional groups based on their hearing frequency range. Table 14 shows the subdivision and thresholds for TTS and PTS from impulsive sounds for these different functional groups. Impulsive sounds include underwater explosions, seismic surveys and pile driving. Since there are no studies on PTS in marine mammals, it is calculated based on the thresholds for TTS. TTS is determined on the basis of known data and is set to the dual threshold of $SPL_{(peak)}$ 196 dB re 1 µPa or $SEL_{(cum)}$ 162 dB re 1 µPa$^2$s (cumulative for the activity over 24 hours) for harbour porpoises and other marine mammals in the functional group that has the best hearing within high frequencies. Calculated PTS ends up at $SPL_{(peak)}$ 202 dB re 1 µPa or $SEL_{(cum)}$ 177 dB re 1 µPa$^2$s for the same functional group.

**Table 14. Threshold levels for TTS and PTS, unweighted. Harbour porpoises belong to the functional group "High-frequency cetaceans". "Source" refers to the sound source. NB = narrow band. From NOAA (2015).**

| | PTS, onset (received level) | | | | TTS, onset (received level) | | | |
|---|---|---|---|---|---|---|---|---|
| | Impulsive | | Non-impulsive | | Impulsive | | Non-impulsive | |
| **Functional group** | **$SPL_{peak}$ (dB re 1 μPa)** | **$SEL_{cum}$ (dB re 1 $\mu Pa^2 s$)** | **$SPL_{(peak)}$ (dB re 1 μPa)** | **$SEL_{cum}$ (dB re 1 $\mu Pa^2 s$)** | **$SPL_{peak}$ (dB re 1 μPa)** | **$SEL_{cum}$ (dB re 1 $\mu Pa^2 s$)** | **$SPL_{(peak)}$ (dB re 1 μPa)** | **$SEL_{cum}$ (dB re 1 $\mu Pa^2 s$)** |
| **Low-frequency cetaceans** | Source: All 230 | Source: All 192 | Source: All 230 | Source: All 207 | Source: All 224 | Source: All 177 | Source: All 224 | Source: All 187 |
| **Mid-frequency cetaceans** | Source: All 230 | Source: All 200 | Source: NB ≥ 3 kHz 230 | Source: NB ≥ 3 kHz 199 | Source: All 224 | Source: All 185 | Source: NB ≥ 3 kHz 224 | Source: NB ≥ 3 kHz 179 |
| | | | Source: All others 230 | Source: All others 212 | | | Source: All others 224 | Source: All others 192 |
| **High-frequency cetaceans (ex: harbour porpoise)** | Source: All 202 | Source: All 177 | Source: NB ≥ 3 kHz 202 | Source: NB ≥ 3 kHz 171 | Source: All 196 | Source: All 162 | Source: NB ≥ 3 kHz 196 | Source: NB ≥ 3 kHz 151 |
| | | | Source: All others 202 | Source: All others 194 | | | Source: All others 196 | Source: All others 174 |

### 7.3.4 Netherlands

In response to the Netherlands' significant investment in renewable energy from offshore wind power, the state agency Rijkswaterstaat developed frameworks for determining the cumulative effects of impulsive sounds on relevant marine mammal populations in the North Sea (de Jong et al., 2015). The agency wanted to develop a method to quantify the potential cumulative effects of impulsive sounds with a focus on harbour porpoises and to attempt to estimate the impact of future wind farm installations. The effects on behaviour (flight) and on hearing (especially PTS) were examined. The threshold values used to determine the effects are presented in Table 15. In the latest version, the threshold for future assessment of environmental impacts was set at $SEL_{(ss)}$140 dB re 1 $\mu Pa^2 s$, the same value used in the German guidelines.

**Table 15. Threshold values for estimating effects on harbour porpoises. $SEL_{(cum)}$ is the cumulative sound that a swimming animal experiences throughout the pile driving. From de Jong et al. (2015).**

| Species | Effect | Threshold | Source |
|---|---|---|---|
| Harbour porpoise | Flight | $SEL_{ss}$ > 140 dB re 1 $\mu Pa^2 s$ | See text below |
| | TTS, onset | $SEL_{cum}$ > 164 dB re 1 $\mu Pa^2 s$ | Lucke et al. 2009 |
| | TTS, after1 h | $SEL_{cum}$ > 169 dB re 1 $\mu Pa^2 s$ | TTS, onset + 5 dB |
| | PTS, onset | $SEL_{cum}$ > 179 dB re 1 $\mu Pa^2 s$ | TTS, onset + 15 dB |

They provide recommendations for guidelines when assessing the environmental impacts of future Dutch projects. They recommend to first calculate the sound's propagation per pile strike around the pile. Based on the calculated sound propagation model and the specified thresholds, the area within which the harbour porpoises are expected to be disturbed is then estimated. The recommended threshold for harbour porpoises is set to an unweighted sound exposure level of $SEL_{(ss)}$ 140 dB re 1 $\mu Pa^2 s$. The threshold is determined as a compromise between the disturbance effects observed in laboratory studies ($SEL_{(ss)}$ 136 dB re 1 µPa²s, Kastelein et al., 2013, referenced in de Jong et al., 2015) and observations in the field ($SEL_{(ss)}$ 144 dB re 1 µPa²s, Diederichs et al., 2014). Preferably two calculations of the disturbance area should be made, one with and one without wind. The next step is to calculate the possible number of disturbed harbour porpoises per pile strike based on the calculated area of disturbance and the estimated population density around the area. They make the assumption that pile driving work using one pile takes up a full day, and so make their calculation using the number of days the harbour porpoises are disturbed during the entire project. Based on the number of disturbance days, the possible effect on the entire harbour purpose population is estimated; they use the Population Consequences of Disturbance model (PCoD). Finally, they also recommend calculating the distance within which there is a risk of PTS to harbour porpoises.

The proposed method was used to estimate the extent of the potential cumulative effects of planned offshore wind farm construction in 2016–2022 in the southern North Sea and the Dutch continental shelf. They test different scenarios, both with and without mitigations. The simulation results in a theoretical reduction in the main harbour porpoise population during the years of active construction when the German threshold of SEL 160 dB re 1 µPa²s was used. They also see an effect of the German threshold that led to a reduction in the number of days of disturbance compared with scenarios without noise mitigation methods which at 750 metres experienced a noise level of SEL ≈ 174 dB re 1 µPa²s.

Prior to construction of the Netherlands' big investment, an environmental impact assessment was made for Borssele and plans for the area were reviewed (van Duin et al., 2015). During the project, they will use a threshold for the entire area. The threshold varies depending on which part of the area construction work takes place in and the time of year. This is because the number of harbour porpoises on the Dutch continental shelf varies over time, with much lower density in the summer and autumn than in the spring. They therefore consider that the thresholds during the summer and autumn do not need to be as restrictive. The value of the threshold varies between the subareas within an area, but the lowest threshold for the time period applies as the threshold for the entire area.

The threshold is a minimum limit of SEL 159 dB re 1 µPa²s at a distance of 750 metres from the sound source. The threshold is the same as the German thresholds of SEL 160 dB re 1 µPa²s but with a safety margin of 1 dB. Past experience has shown that it can be difficult during construction start-up to maintain the threshold and that the effect of mitigation methods can decrease in certain circumstances (e.g., bad weather conditions). A lower value allows some variation without the sound exceeding the set target of SEL 160 dB re 1 µPa²s. The lowest value applies to periods with the most harbour porpoises in the area, between January and May. The threshold varies throughout the year and depending on sub-area, with a maximum value of SEL 172 dB re 1 µPa²s during the period September to December.

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# A framework for regulating underwater noise during pile driving

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During the construction of bridges, offshore wind farms and other offshore or near-shore structures, some form of pile driving method is often used to drive the structure into the bottom. This may cause noise levels that are so high that marine organisms can be disturbed, harmed or even killed.

This study has produced a scientific basis for assessing underwater pile driving noise and its effects on marine life. The report includes technical descriptions of pile driving activities, underwater acoustics, sound propagation, and the impact on harbour porpoises, the fish species cod and herring, fish eggs and fish larvae.

Today, Sweden lacks established thresholds for when underwater noise poses a threat to marine animals. The authors propose harmful levels for injury and negative effects, which can then be used to establish guidance values for regulating underwater noise that are adapted for Swedish waters and species. Several European countries have some form of thresholds indicating when serious environmental impacts can occur, as well as standards for measuring and reporting underwater noise.

**The Vindval research programme** collects, creates and communicates information and facts about the environmental impact of wind power on the environment, the social landscape and people's perception of wind power installations. Vindval provides funding for research, including literature reviews and syntheses regarding the effects and experiences of wind power.

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